Report Prepared for the Research Division
Royal Commission on National Passenger Transportation

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EXTERNALITY PRICING

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I. Introduction

This study reviews the pros and cons of externality pricing as it relates to the passenger transportation sector. Its focus is on approaches to control of air pollution from the automobile and noise pollution from airplanes, which are the main subjects of the academic literature in this area.

The study begins with a comprehensive survey of the literature pertaining to externality pricing and its proxies. The following section investigates the proposed and actual uses of externality pricing and an analysis of second-best prices and other economic incentive schemes[1] which have been popularized in the externality literature.

The purpose of externality pricing is to correct a resource misallocation. The misallocations, or market failures, dealt with in this study result from externalities. As Baumol and Oates (1988) point out ". . . An externality is present whenever some individual's (say A's) utility or production relationships include real (that is, non-monetary) variables, whose values are chosen by others (persons, corporations, governments) without particular attention to the effects on A's welfare" (p. 17).

Examples of an externality occur when:

a) a driver takes his car to work in rush-hour traffic thus imposing a time cost on other drivers on the road;

b) a jet aircraft takes off from Dorval Airport in the evening, waking residents who live near the airport;

c) A thermal power plant emits sulfur dioxide from its smoke stack which injures vegetation and lakes down wind from the plant.

These externalities (or more precisely, external diseconomies) involve actions by agents which impose costs on others. The market provides no incentive for agents to take these costs into account. Thus as Baumol and Oates (1988) point out, for this externality to be considered a misallocation it is also necessary that ". . . [t]he decision maker, whose activity affects others' utility levels or enters their production function, does not receive (pay) in compensation for his activity an amount equal in value to the resulting benefits (or costs) to others". (pp. 17–18)[2]

Economists also refer to such misallocations as market failures, since agents undertake actions (such as reducing the availability of clean air or peace and quiet) which use up scarce resources without incorporating the costs of such actions into their decision calculus. The economic solution to such problems involves the setting of implicit or administratively determined prices, sometimes referred to as externality prices, which internalizes these external costs.[3]

Sections II, III and IV of the report explore the theoretical strengths and weaknesses of externality pricing including a discussion of issues such as:

a) why a large segment of public policy-makers favour direct regulation of externality problems rather than pricing schemes; and,

b) why economists feel that externality pricing (of some sort) or at least some form of economic incentives are superior to direct controls.

Finally, section V discusses the possible role of externality pricing in alleviating transportation air and noise pollution problems.

II. Theoretical Justification of Externality Pricing

The source of externality problems is often found in poorly defined property rights. This can be demonstrated with a simple example of an environmental problem. The example also provides a theoretical basis for an effluent charge or externality price resulting from the failure of the market to place a price on environmental services (see Anderson et al., 1977; Baumol and Oates, 1971; Kneese, 1977; Kneese and Schultz, 1975; Sims, 1979).

The environment is an asset which yields waste-assimilative services to users, for example pulp and paper mills, municipal sewage treatment plants, smelters, etc. Problems arise to the extent that the asset is scarce. More services to firms, that is, more waste emissions by firms, means less environmental quality and hence less recreational and other services available to other users. The failure of the market to reflect this opportunity cost, or these damages, in a price for these waste-assimilative services, induces their overuse by polluters. When making their production decisions, firms should internalize the damages which result from their pollution emissions, otherwise they would underestimate the costs of production and hence over-produce. A charge per unit of pollution equal to the social damage of that unit of pollution would solve the environmental problem. This is the basis on which effluent charges (EC) or externality prices were initially suggested.[4]

Technically this problem can be investigated in a simple partial equilibrium model.[5] Assume there are two dischargers of the same pollutant which causes damage at a single receptor point. e_i is the total emissions per unit of time from source i. In an uncontrolled state, source i would emit Z_i. If this polluter emits less than Z_i it is abating pollution. Abatement at source i (A_i) is defined as:

$$A_i = Z_i - e_i \qquad i = 1, 2. \qquad (1)$$

Each source finds it costly to reduce pollution below Z_i and thus in the absence of some sort of external pressure, or regulation, will chose not to abate.[6] The cost of abatement is defined as:

$$TC_i^A = C^i(A_i) = C^i(Z_i - e_i) \qquad (2)$$

where $$\frac{dC^i}{dA_i} > 0 \text{ and } \frac{d^2C^i}{dA_i^2} \geq 0$$

The damage TD, suffered at the receptor, are defined as:[7]

$$TD = f(e_1 + e_2) = f(e) \qquad (3)$$

where $$\frac{df}{de} > 0 \text{ and } \frac{d^2f}{de^2} \geq 0$$

The regulatory agency's goal in this situation is to minimize the sum of the damage and abatement costs, i.e.

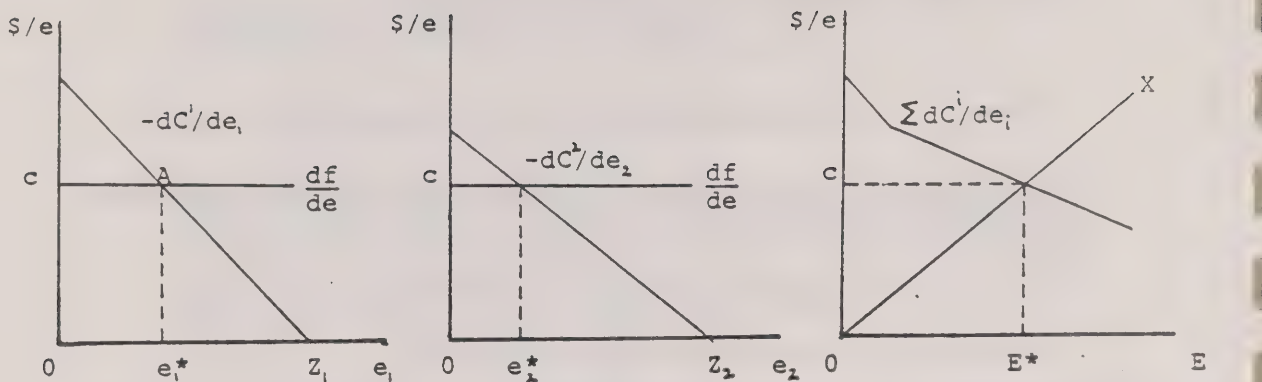
$$\min f(e_1 + e_2) + C^1(Z_1 - e_1) + C^2(Z_2 - e_2) \qquad (4)$$

The solution to this problem yields the following results:

$$\frac{df}{de} = \frac{dC^1}{dA_1} = \frac{dC^2}{dA_2} = c \qquad (5)$$

where c is the constant marginal damage of pollution emissions. A graphic representation of this solution is provided in Figure I, diagrams (a) and (b). The optimal externality price is shown to be equal to c, the marginal damage of emissions. This will result in the pollution sources choosing to emit e_1^* and e_2^* and thus abating $Z_1 - e_1^*$ and $Z_2 - e_2^*$, respectively. Pollution source 1 will pay ce_1^* for the emissions that continue. This is equal to the total damages of e_1^* emissions. Note overall damages are $c(e_1^* + e_2^*)$ which is equal to the total charge payment by the two polluters. As a result, the polluters are induced to take the damages they impose into account in their production decisions.

FIGURE I
THE OPTIMAL EXTERNALITY PRICE



a) Pollution Source 1 b) Pollution Source 2 c) Aggregate Pollutio

Note: $TC_A^i = C^i(Z_i - e_i)$ $i = 1, 2$

$$\frac{dC^i}{de_i} = \frac{dC^i}{dA_i}\frac{\partial A_i}{\partial e_i}$$

Therefore $-\frac{dC^i}{de_i} = \frac{dC^i}{dA_i}$

Concentrating on point source 1, it is easy to see why abatement does not exceed $Z_1 - e_1^*$. Any abatement beyond this point would involve a marginal cost of abatement which is larger than both the damages created by additional emissions[8] and the externality price, c. If the authority knows the marginal damages of emissions and if they are constant as has been assumed to this point, it need only set a charge c. The result will be optimal as the firms will automatically adjust their emissions to e_1^* and e_2^*. The authority need have no knowledge of abatement costs.[9]

An alternative regulatory device which is certainly more prevalent in North America is the effluent standard. Optimality in the case just examined would involve setting an effluent standard of e_1^* per unit of time for pollution source 1 and e_2^* per unit of time for pollution source 2. The determination of these standards would require knowledge of not only the level of marginal damage, but also the marginal cost functions for all polluters. Thus there is an obvious informational advantage of the externality price scheme over the effluent standard.[10]

It thus appears that charges have a substantial advantage over the effluent standard or direct regulation[11] approach. The apparent advantage is, however, neutralized to some degree by the introduction of various assumptions which introduce a greater degree of realism into the model.

III. Externality Pricing Under Alternate Assumptions

A. Nonlinear Damage Functions

To this point, the damage function has been assumed to be linear. This yields a constant marginal damage function and rules out, for example, threshold affects of pollutants. However, there is no reason to expect that damage functions will be linear. Indeed, the ultimate shape of a pollution damage function depends on various parameters and is ultimately an empirical question.[12]

The advantage of the externality price discussed above is at least partially neutralized when the damage function is not linear. Such a situation is depicted in Figure 1(c). The downward sloping, kinked curve is the marginal abatement cost function. The curve labelled X is the new marginal damage of emissions function. The optimal level of abatement is $e^* = e_1^* + e_2^*$ and the optimal externality price is still c. Optimality now requires that:

$$\frac{df}{de}(e^*) = \frac{dC^1}{dA_1}(A_1^*) = \frac{dC^2}{dA_2}(A_2^*) \tag{5'}$$

Given that the authority knows the function df/de, it must first determine e^* before setting the optimal externality price, c. The problem is that it is no longer precisely true that the optimal charge equals marginal damage. The precise statement is that it equals marginal damage at the optimal level of emissions (e^*). The determination of e^* would appear to require knowledge of marginal damage and all marginal costs. The information advantages of the charge appear to have vanished.[13]

B. Spatially Differentiated Damages

The previous two sections have presented models in which a single externality price for a given pollutant is optimal. With a slight extension to the models, it can be shown that there are likely many real-world situations for which this will not hold. The extension involves the assumption that the location of the polluter is relevant to the damage created by its emissions of a given pollutant. For example, a polluter located downwind (or downstream) from a receptor will, on average, have no impact on the air quality (or water quality) and hence the pollution damage at that receptor point. Obviously, the same is not true of an identical polluter located upwind (or upstream) from the receptor.[14]

The relevance of this point can be demonstrated with the following model of a hypothetical airshed. Assume a region can be divided into n areas of homogeneous air quality or receptors with m polluters located throughout. The damage or pollution, in each area (n), is a function of air quality, Q_i. Thus,

$$D_i = D_i(Q_i) \qquad i = 1, 2, \ldots n.$$

The total regional damages are thus:

$$D = \sum_{i=1}^{n} D_i$$

The cost of abatement to society is:

$$C = \sum_{j=1}^{m} C^j \ (A_j)$$

The air quality at receptor i is:

$$Q_i = Q_i(e_1, e_2, \ldots e_m)$$

where e_i is emissions from source i.

Minimizing the costs of pollution implies:

$$\text{Min } D + C = \sum_{i=1}^{n} D_i(Q_i(e_1,\ldots e_n)) + \sum_{j=1}^{m} C^j\ (e_j)$$

$$\sum_{i=1}^{n} \frac{\partial D_i}{\partial Q_i}\frac{\partial Q_i}{\partial e_j} = -\frac{dC^j}{de_j} \qquad j = 1, 2,\ldots m. \qquad (5'')$$

This yields a system of m equations with m unknowns. If the damage function is linear in quality, then $\partial D_i/\partial Q_i = \alpha_i$. Thus optimality requires that,

$$\sum_{i=1}^{n} \alpha_i \frac{\partial Q_i}{\partial e_j} = -\frac{dC^j}{de_j} = c_j \qquad (5''')$$

where c_j is the jth source's marginal cost of abatement.

For most types of polluters there is no reason to expect that a unit of emissions from all j sources will affect air quality in region i in the same manner. Thus $\partial Q_i/\partial e_j$ will vary by the location of the polluter. This means that each polluter must now be assigned a different externality price.

Simulation models (Krupnick, 1986; Seskin, Anderson and Reid, 1983; Oates, Portney and McGartland, 1989) are used to investigate questions such as what is the cost of imposing a uniform externality price when the optimal price should vary by the location of the polluter? Such studies traditionally use meteorological models which yield constant values for $\partial Q_i/\partial e_j$. If

$$\frac{\partial Q_i}{\partial e_j} = a_{ij} \qquad (6)$$

then equation (5''') becomes,

$$\sum_{i=1}^{n} \alpha_i a_{ij} = c_j$$

The authority can still set the optimal charge provided it knows the damage function and the impact of discharges on air quality. Thus the authority once again needs no information on the marginal costs of abatement. While the externality price (c_j) must vary across sources, it retains the informational advantages originally attributed to it.

Once again it should be clear, however, that this advantage only occurs if the damage function is linear. If it is assumed that the damage function is not linear then condition (5‴) becomes,

$$\sum_{l=1}^{n} \frac{\partial D_l}{\partial Q_l} (Q_l)\, a_{lj} = - \frac{dC^j}{de_j} \qquad j = 1,...m.$$

Since $Q_l = Q_l(e_1, e_2, ...e_m)$, the optimal charge is no longer independent of the optimal discharge level. Thus the above m equations must be solved simultaneously and require knowledge of abatement costs as well as damages and meteorological conditions.

It is unrealistic to assume that in real-world situations the damage function will always be linear, or that the authorities can easily generate a damage or marginal damage function (Dales, 1968). As a result, many economists have abandoned the idea of an optimal externality price for various second-best pricing schemes.

C. Uncertainty

While the analysis to this point suggests that externality problems will not easily be solved with the use of pricing, it has provided no insights into the popularity of the direct regulation (or command-and-control) approach which dominates much of our environmental regulation. Uncertainty on the part of regulatory authorities provides evidence of situations in which standards may dominate charges.

A simple model based on the work of Weitzman (1974), Adar and Griffin (1976), Fishelson (1976), Roberts and Spence (1976) and Watson and Ridker (1984) demonstrates such situations. To simplify the analysis, assume that the airshed in the model is perfectly mixed so that the location of the polluting firm and receptor of the pollution is irrelevant. Also assume that the marginal abatement cost curve and the marginal damage curve are linear.[15] The analysis generally compares the relative welfare losses imposed on society by an externality price and a quantity-based regulation when the environmental authority is uncertain about marginal damages or marginal abatement costs.

Figure II (a) shows a situation in which the authorities have underestimated the true marginal damages of emissions (MD_e^T). Under a standard scheme they allow emissions of e, which is greater than the optimal level e^*. Supra-optimal emissions increase damages equal to the area e^*ACe, which is partially offset by a gain to polluters in terms of reduced abatement costs equal to the area e^*ABe. The social loss from the standard scheme is the area ABC. With a pricing scheme the set charge, EC, is less than the optimal externality price, EC^*. This leads to emissions of e rather than e^* and a social loss equal to that under the standard. Thus,

uncertainty with respect to the marginal damage function provides no advantage to either scheme.

In the case of uncertainty with respect to marginal abatement costs, this symmetry no longer holds. Assume that, as in diagram (b) of Figure II, the true marginal cost of abatement (MC_A^T) is underestimated. In a standard scheme, emissions of e would be allowed which is less than the optimal level e^*. This will reduce damages by $eACe^*$ but will increase abatement costs by the amount $eBCe^*$. The net social loss from the standard scheme is thus ABC.

Under a pricing scheme, the authorities would set a charge of EC which is less than the optimal externality price of EC^*. This would result in e' emissions, which is larger than e^*. This would increase pollution damages by e^*CDe' and reduce abatement costs by e^*CEe'. The net social loss from the pricing scheme is CDE.

In general there is no reason to expect CDE and ABC to be equal. Adar and Griffin (1976) show that:

$$WL_T - WL_q = -\frac{1}{2}(EC/e)(\Delta e)^2\left[\frac{1}{\varepsilon_D} + \frac{1}{\varepsilon_c}\right]$$

where WL_T is the welfare loss from the pricing scheme (area CDE in the previous example); WL_q is the welfare loss from the standard scheme (area ABC in the previous example); ε_D is the elasticity of the marginal damage function; ε_c is the elasticity of the marginal cost of abatement function; and $\Delta e = e' - e$.

Thus, the welfare loss from charges or standards will be equal if:

(i) the damage and cost elasticities are equal in absolute value; or

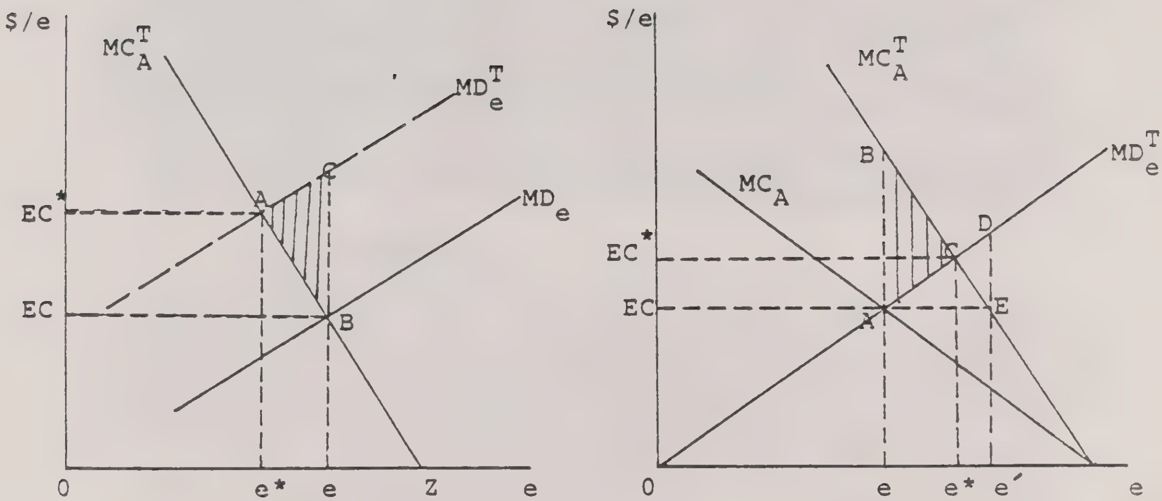
(ii) Δe is 0, i.e. $MC_A = MC_A^T$, the authority knows the true marginal cost of abatement.

Standards are preferable to charges when $WL_T - WL_q > 0$, which occurs if

$$|\varepsilon_D| < |\varepsilon_c|$$

Thus, if $|\varepsilon_D| \Rightarrow 0$, standards are preferable, whereas if $|\varepsilon_c| \Rightarrow 0$, taxes are preferable. The rationale for this result is as follows:

FIGURE II
PRICES VERSUS QUANTITIES WITH UNCERTAINTY



a) Damage Uncertainty

b) Cost Uncertainty

a) If the marginal damage function is steep, as might be the case for certain very noxious pollutants, even a slight error in emission levels will result in large damages. With uncertainty about costs, the chances of such errors are greater with a pricing scheme.

b) If marginal damages are relatively flat, a charge will better approximate marginal damages. Indeed, as has already been shown above, with a linear damage function, the charge can lead to an optimal result independent of any knowledge about costs.[16]

c) If marginal costs are steep, then an overly ambitious standard could result in excessive costs to abators. The charge places an upper limit on such costs.

The key characteristic in these two schemes is that charges set an upper limit on costs, whereas standards set an upper limit on discharges.[17]

Watson and Ridker (1984) extend this analysis to nonlinear control cost and damage functions, as well as multiplicative, nonsymmetric errors for both functions. The results of this analysis are consistent with those of Adar and Griffin (1976). Watson and Ridker find that selective use of effluent charges at various U.S. sources of different pollutants could lead to savings of about $175 billion between 1975 and 2025.[18]

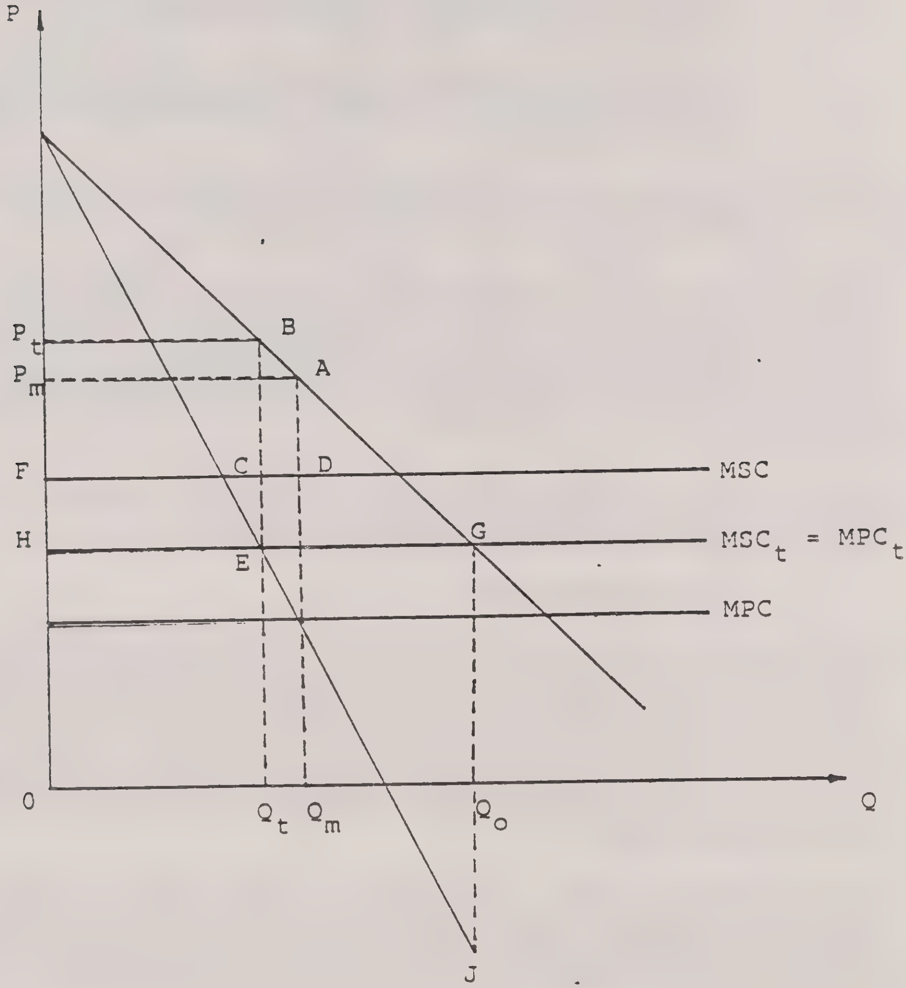
Nevertheless the introduction of uncertainty into the analysis suggests that the superiority of externality pricing in all cases is questionable.

D. Market Imperfections

If we ignore these previous problems and assume a linear damage function, it may still be shown that the "optimal" externality price discussed earlier is not desirable. Implicit in the earlier analysis was the absence of other market failures. If the polluting firm is a monopoly (or, to take a less extreme example, has market power) an externality price set equal to the constant marginal damage of pollution will not yield an optimal result (Buchanan, 1969; Oates and Strassmann, 1984; Baumol and Oates, 1988).[19]

Figure III illustrates this result. The monopolist produces subject to a constant marginal cost (MPC) and chooses to produce the profit-maximizing output level OQ_m for which it charges a price of OP_m. Assume that the environmental authority decides to impose a charge on the firm for each unit of pollution it emits to the

FIGURE III
THE IMPACT OF EXTERNALITY PRICING ON A MONOPOLY



environment.[20] This induces the polluting firm to reduce its pollution emissions per unit of output, which in turn causes the rise in the marginal private cost of production. Since emissions per unit of output are reduced, then the marginal social cost of output is lowered. If the charge is set equal to marginal damages so that the externality is internalized then the new marginal social and private costs must be equal, i.e. $MSC_t = MPC_t$. As a result, output will fall to Q_t and price will rise to P_t.

This policy results in a cost savings to society, in the form of reduced pollution damages, equal to the area FCEH. As well, the reduction in output from Q_m to Q_t results in a welfare loss of ABCD.[21] Whether the externality price or Pigouvian tax[22] will lead to an improvement in welfare depends on the relative sizes of areas FCEH and ABCD.

Oates and Strassmann (1984) suggest that from available parameter values, the gains from environmental improvement (area FCEH) will far exceed the losses from reduced output (area ABCD).

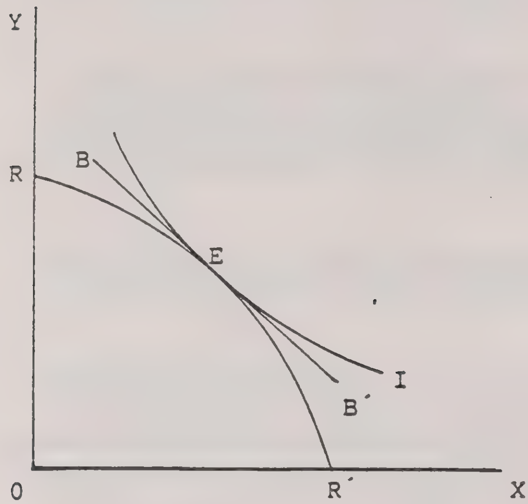
Another concern related to market structure is the reaction of firms to an optimal charge. The polluter, through appropriate actions, may be able to affect the charge. If the authority sets an optimal charge and then does not adjust it, there is no problem. However, as is pointed out by Baumol and Oates (1988) if the tax is set iteratively,[23] as was discussed above in section IIIA, by continuously setting the charge equal to "current" marginal damages, problems may arise. If the polluting firm realizes it can affect the charge by altering emissions,[24] the result of an otherwise optimal tax could be supra-optimal emissions.

E. Nonconvexities

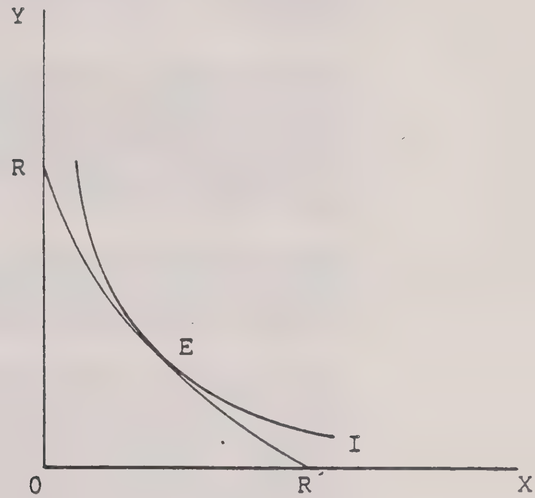
Baumol and Oates (1988) show that a sufficiently strong externality, which interferes with the production of some other good, could give rise to a nonconvexity in the social production set. In such a case, the second-order conditions for a social optimum will be violated. If this occurs, society may be left with the difficult problem of choosing among a number of local optima.

When the second-order conditions are fulfilled and a competitive equilibrium (in the absence of any market failure) attained, the economy will achieve a Pareto optimal allocation of resources. At this allocation all producers and consumers will be in equilibrium and the value of total output will be maximized.[25] A typical Pareto optimal solution is shown in Figure IV(a). In this diagram, RR′ is the production-possibilities curve; I is the community-indifference curve; and BB′ is the budget line. A decentralized economy under the conditions described above will attain an equilibrium at E. At this point, a Pareto optimal solution has been attained and the value of output has been maximized. Indeed it is because these

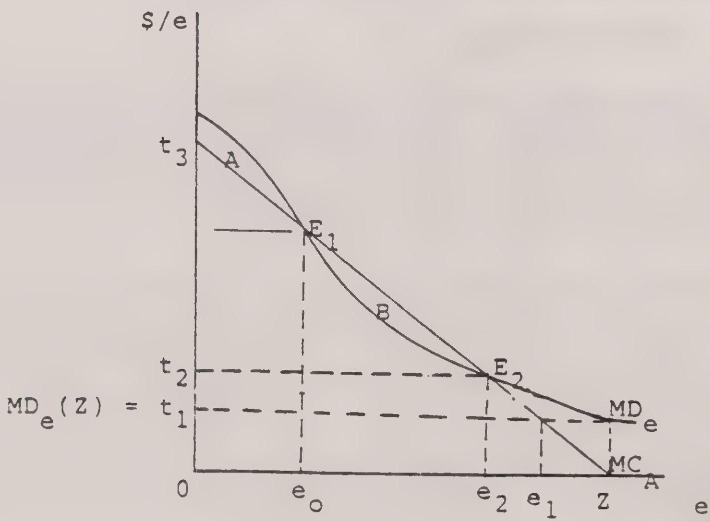
FIGURE IV
NONCONVEXITIES AND PARETO OPTIMALITY



a) A Pareto Optimal Solution



b) Nonconvexities



c) Diminishing Marginal Impact of Pollution

two results coincide and because a competitive economy maximizes the value of output, that we can attribute normative significance to a competitive outcome.[26]

The introduction of nonconvexities, however, creates difficulties. Assuming a strictly convex production-possibilities curve, as in Figure IV(b), an interior point can no longer represent the position at which the value of output is maximized. Indeed there will be two local value maxima at R and R′. If the price of X is high relative to Y, the highest value of output will be attained at R′ and vice versa for a relatively high value of Y. However, it does not follow that the social optimum must lie at either R or R′ as is shown in diagram (b). In this case the social optimum, at E, cannot correspond to a maximum output value.

Thus, we can no longer be certain that a Pigouvian tax, or externality price, which is designed to maximize the value of output, will lead the economy to a Pareto optimal outcome. As is pointed out by Baumol and Oates (1988), this tax might not even lead the economy in the right direction. The violation of the standard concavity-convexity conditions, which is more probable in the presence of externalities, leads to multiple local optima (which fulfill the first-order conditions for an optimum). Even if a policy (such as an externality price) can be depended on to lead the economy to a local optimum there is no guarantee that this will be the global optimum.

Burrows (1980) provides a simple explanation of this phenomenon. While it is generally expected that the marginal damage of pollution will rise (or be constant throughout), that may not always be the case. For example, if pollution drives the receptor from a region because of damages suffered, thereafter the additional damages of increased pollution would be 0. Thus, if there is diminishing marginal impact of pollution,[27] the marginal damages of pollution may eventually diminish. This could lead to a situation like that depicted in Figure IV(c).

There are two local optima depicted, E_1 and E_2, at which the first-order conditions are satisfied. Assume that the economy is initially in the uncontrolled state with $e = Z$ and that the authorities chose an iterative pricing approach. It first calculates the current emissions, Z, and the current marginal damages $MD_e(Z)$ and sets a tax equal to those damages, i.e. t_1. The polluters would react by reducing emissions until $t_1 = MC_A$, which occurs at e_1. The authority would then adjust the tax to $MD_e(e_1)$ and the process would continue. It would eventually stop at E_2, one of the local optima. But is this the global optimum? As can be seen in Figure IV(c), if area A exceeds area B, the global optimum could occur at $e = 0$.[28] Thus, when the marginal impact of pollution diminishes after some point (which is exactly the case considered in Baumol and Oates) the iterative procedure suggested in Baumol (1972) may lead the economy to the wrong optimum. Of course if the authority had complete information on MD_e and MC_A, it could set an appropriate and optimal externality price. If area A is larger than area B, the optimal tax is t_3,

whereas if B > A it is t_2. Once again, however, the apparent informational advantages of the optimal tax vanish.

F. Summary

There are a number of reasons to doubt the ability of a regulatory agency to use theoretically optimal externality pricing. These include nonlinearity of the damage function, spatially differentiated damages, uncertainty with regard to the true marginal damage and abatement cost curves, market power and nonconvexities. The most damaging of all these is the last. It implies that we cannot even be sure of the correct direction in which to modify an externality.[29] This problem, however, probably affects all control policies equally.

Another key point is that with current levels of knowledge we are unable for most pollution problems to say much about the location or level of the marginal damage of pollution function (Baumol and Oates, 1988). This is the ultimate Achilles heel of the maximizing approach to environmental policy.[30] As a result, much of the current literature on externality pricing has lowered its sights somewhat by adopting a "satisficing" approach. First discussed in detail by Baumol and Oates (1971), this approach uses standards set by the government agency as targets and adopts the most cost-effective policy available to achieve these goals. `It is based on an implicit separability of decisions on the optimal policy goal and the ideal policy[31] to achieve it, as well as a belief that the true comparative advantage of economics is in the determination of the latter decision.

The externality price has been resurrected by some economists as the policy of choice to achieve the goals of the "satisficing approach." The next section of this study looks at this approach and presents arguments for and against the use of second-best externality prices to achieve environmental goals.

IV. The "Satisficing" Approach to Externality Pricing

A. Introduction

Much of the current environmental economics literature incorporates two main themes:

a) direct regulation versus economic incentives; and

b) the choice of economic incentives, for example, effluent charges versus transferable discharge permits (TDP).

This section discusses the advantages and disadvantages of economic incentives (primarily second-best externality prices or effluent charges). The case made for or against these incentives is relative rather than absolute since ultimately, we are not asking which policy will lead to a Pareto optimal solution, but rather which policy is better than the others. Which policy will achieve a given standard (that may or may not be optimal) in the most desirable way. The evaluation of these policies is based on concepts such as economic efficiency, dynamic efficiency, informational requirements and political acceptability. Ultimately, no policy will prove superior on all counts, and some situations call for a mixture of policies. Nevertheless, there appears to be significant evidence supporting some form of economic incentives to supplement, if not replace, direct regulation in certain situations.

To place the analysis of economic incentive schemes in the proper light, it is necessary first to look at the form of environmental regulation currently used almost exclusively throughout North America: direct regulation.

B. The Direct Regulation Approach

Jurisdictions throughout North America are committed to a direct regulation-enforcement framework to control environmental problems. Ambient and effluent standards[32] provide objectives which are, at least in theory, enforced by the threat of fines and/or imprisonment.[33] This framework often includes some form of financial assistance which is intended to help the polluter comply with environmental objectives and regulations. The most popular types of financial assistance in North America include accelerated depreciation allowances on capital expenditures for pollution control, the refund of sales taxes on the purchase of pollution control equipment and direct subsidies and loans for the purchase of pollution control equipment.

The approach to environmental regulation used in Ontario and in other jurisdictions has been described as one of "symbolic actions" (Dewees, 1980). Tough standards which sometimes appear to ignore abatement costs are established in law or in regulations. They are usually founded on the protection of human health. But the apparent rigidity of these standards is only symbolic. When setting objectives, deadlines and enforcing compliance, exceptions are made through bargaining and negotiations. Agreements which may be less restrictive than the letter of the law are made. Implementation deadlines are often deferred. Enforcement through court action and fines is viewed only as a last resort.[34]

One of the chief criticisms of the direct regulation-enforcement framework is that it has not been successful in persuading polluters to comply with pollution abatement schedules developed by environmental authorities. It has been argued that this method of regulation, instead of providing an incentive to abate, has provided an incentive for polluters to delay expenditures on pollution abatement (Roberts, 1970;

Mills and White, 1978; Nemetz, 1986; Tietenberg, 1988). In deciding whether to comply with current environmental requirements, a firm will weigh the cost of noncompliance or delay against the "rewards"--the cost savings because research and development may ultimately produce cheaper methods of abatement, more generous financial assistance from governments in the future, and the use of scarce capital that would have gone to purchase pollution control equipment for profit-making investments. In addition, noncomplying firms save on operating and maintenance costs which would have been incurred had the appropriate abatement equipment been in place.

The costs of noncompliance, or delay, are the adverse public reaction and fines which result from prosecution. Given the technical complexity of many modern production processes, however, it is easy to give the appearance of co-operation by undertaking successive engineering and cost studies. This also helps highlight technical difficulties which can cause further delay and help to avoid prosecution. In essence, the direct regulation-enforcement approach provides significant rewards for noncompliance and provides insignificant costs from delay.

Thus, if a firm does not comply with environmental standards, it runs the rather small risk of prosecution or fines. But it is unlikely that the magnitude of the fine will exceed the cost savings from delaying compliance. In addition, a firm which exceeds a standard must first be caught which, if it is one of many firms, is problematical. Even if it is caught, it can probably negotiate if, as in most jurisdictions with finite budgets for environmental litigation, the authority is seeking voluntary compliance. Moreover, where abatement options are costly and uncertain, the firm can make a good case regarding the unreasonableness and infeasibility of the current requirements. Any delay, whether this delay be the result of litigation or bargaining with the environmental agency, has clear and substantial benefits to the polluting firm.

By reducing the cost of abatement, financial assistance tends to reduce the reward to polluters from delaying compliance. But as long as abatement costs remain a substantial net loss item for the firm (as it likely will even with significant tax-incentive and subsidy schemes) it is unlikely that financial assistance will persuade firms to comply.

In the existing literature there is ample evidence of this tendency to delay compliance or only to appear to comply. A case in point is INCO. A Ministerial Order issued in 1970 required a staggered reduction of INCO's sulfur dioxide (SO_2) emissions at Copper Cliff to the following levels:

1 July 1970	5200 tons per day
31 December 1974	4400 tons per day
31 December 1976	3600 tons per day
31 December 1978	750 tons per day

Originally INCO had intended to use a hydrometallurgical process to remove about 40 percent of the sulphur from the total feed to the smelter (Felske, 1981). In the end, INCO refused to implement this process. In 1973, INCO requested a two-year extension for the 1976 limitation, that is, 3600 tons per day, and the elimination of further requirements. In the requested amendment, INCO indicated that ". . . the hydrometallurgical option had under research scrutiny proven to be technically possible but not economically viable" (Felske, 1981). This amendment was denied. In 1975, INCO proposed to reduce emissions through smelter modifications and the production of sulfuric acid. Once again INCO backed out of this commitment for economic reasons. Between 1973 and 1978, INCO conducted a voluntary emission-reduction program based on local meteorological conditions, dispersion modelling and smelter process adjustments. The objective was to control local ground-level concentrations of SO_2. In July 1978, the Ontario Ministry of the Environment issued a Control Order extending an emission level of 3600 tons per day, i. e. the 1976 limitations, to mid-1982.

In 1980, INCO proposed an improved flotation method for increasing pyrrhotite rejection, to be implemented within two years. In April of the same year, the Ministry announced a draft Control Order calling for an immediate limit of emissions to 2500 tons per day with a limit of 1950 tons per day by the end of 1982. These limits were subsequently incorporated in a regulation under the *Environmental Protection Act.*

Felske (1981) suggests that INCO has avoided the constraints imposed in various Control Orders by using technical and economic arguments. The Company promises to adopt technically advanced methods of abatement but in the end refuses to adopt them for economic reasons. The catch is that it is never economically viable to adopt an investment that does not generate revenue.

This opportunity for delay seems to be behind the lack of progress in pollution control by some pulp and paper mills in Ontario. Victor and Burrell (1981) found that while there has been environmental improvement at the industry level, such improvement was slower than expected and did not reflect a uniform pattern among all mills. The ineffective enforcement by the Ministry is dramatically exemplified by the record of prosecutions and fines brought against mills, and the history of Control Order Amendments and postponements until 1977, in Ontario.

There were 17 separate, environmentally-related prosecutions of pulp and paper mills between 1968 and 1977. When fines were imposed they were generally $2000 or less (Victor and Burrell, 1981), an amount unlikely to induce timely compliance among recalcitrant polluters.[35]

Because of the limited success of the regulatory program in use since 1965, the Ontario Ministry put all noncomplying mills on Control Orders in 1977. However, since then, there has been an abundance of Control Order Amendments and

postponements in response to requests by the mills (Victor and Burrell, 1981). This evidence is entirely consistent with Roberts' (1970) suggestion that the direct regulation-enforcement framework provides an incentive for industries to delay expenditures on pollution abatement.

As further evidence of the failure of the direct regulation-enforcement framework in Ontario, the following facts are noted by Victor and Burrell (1981):

(i) Between 1973/74 and 1975/76, the number of mills in Ontario in compliance with federal toxicity requirements rose from one to nine and remained unchanged to 1978. This number represents only about one third of the pulp and paper mills in Ontario. (p. 28)

(ii) In 1965, the Ontario Water Resources Commission sent a directive to all pulp and paper mills in Ontario requiring that the level of suspended solids in waste emissions be reduced to 50 mg/L by December 31, 1966. By 1978, the average discharge from the industry in Ontario was nearly 110 mg/L of suspended solids, nearly double the level required by 1966. It should be noted that theindustry has progressed more quickly with the abatement of suspended solids than with the control of BOD or toxic elements (p. 128).

When regulatory objectives which are more than 15 years old have still not been met, it seems reasonable to speculate that changes are in order.

The history of efforts to control automobile pollution in the U.S. is also one of delay. Indeed in 1969, the U.S. Justice Department brought charges against auto manufacturers for collusion in delaying the development of pollution control equipment.

The first legislated national automotive control standards in the U.S. were passed in the 1965 *Clean Air Act*. This placed restrictions on hydrocarbon (HC) and carbon monoxide (CO) emissions,[36] which were to be met by 1968. The first requirement for mandatory exhaust control devices was passed in California and required installation by the 1966 model year. This was vigorously opposed by manufacturers.[37]

In 1970 the *Clean Air Act* Amendments required that all new car emissions be reduced 90 percent below uncontrolled levels for HC and CO by 1975 and for nitrogen oxide (NO_x) by 1976.

The auto manufacturers requested an extension of these standards by one year. They eventually won this extension in a court case on the grounds of technical infeasibility.[38] A further extension of one year was granted by Congress in 1974

as the result of the OPEC oil embargo. The original 1975 standards had not been met by the early 1980s.

In addition to problems similar to those outlined above with respect to pulp and paper and INCO, there is an additional peculiarity in the history of auto pollution control: the sanctions were so brutal as to make enforcement nearly impossible. If an engine family failed its certification test, the relevant vehicle class could not be sold. Given the importance of the auto industry to the economy, along with the tough foreign competition, such a sanction was unlikely to be stringently imposed on U.S. manufacturers.[39] The regulatory authority was pressed to allow delays, set weaker standards and reduce certification requirements. Eventually, deadlines became so flexible no manufacturer could fail to achieve them.

While the Control Order Amendments and postponements mentioned above may have legitimately resulted from technical difficulties, this does suggest another weakness in the direct regulation-enforcement framework: it does not provide an incentive for technical innovation or advancement. Research and development could overcome the technical difficulties that have necessitated these amendments and postponements. Unfortunately, it is not in the interest of polluters to overcome these difficulties or to expend resources in attempting to do so. In addition, any advancement from such research might be imposed on the firm when Control Orders are renewed. This is not likely to be desirable from the firm's point of view, since it would increase the firm's capital outlay with no corresponding increase in profits. Thus, there is little incentive for polluting firms to conduct serious research into pollution abatement techniques under the direct regulation-enforcement approach.[40]

The existing literature recognizes that under the direct regulation-enforcement approach there is a tendency to set stricter quality standards for new sources than for existing sources. This makes economic sense where new plants can be built to take advantage of economies of scale or install less pollution-intensive processes. This differentiated regulation, however, provides an incentive for firms to continue to operate obsolete facilities, rather than to replace them with new facilities. In this way firms can delay compliance with the more stringent environmental standards which apply to new facilities. Such a policy can, over time, have an adverse impact on productivity and, in the short run, could actually increase pollution emissions.

Gruenspecht (1982) investigated the impact of this grandfathering[41] in the auto industry. Raising the price of new automobiles through stricter controls prolongs the life of older, more emission-intensive automobiles. The possible short-run effect is increased pollution emissions.

Gruenspecht (1982) concludes that while ". . . tighter new source standards are ultimately reflected in lower aggregate emissions, their impact on investment plans

can result in undesirable short-run outcomes. Clearly, other regulatory tools are needed as substitutes for or supplements to the differentially stringent regulation of new sources. Policies designed to promote the retirement of old capital are particularly attractive." (p. 330)

Within the direct regulation-enforcement framework, one of the major sources of financial assistance available to firms has come from tax provisions intended to encourage expenditures on pollution abatement equipment.[42] This type of financial assistance may reduce efficiency: because the tax concessions can be applied only to capital expenditures, they encourage a firm to adopt abatement techniques which need a lot of equipment, even if these techniques are not economical for reducing pollution. An example of the perverse incentive created by such tax provisions is presented in Roberts (1970):

> Suppose . . . a firm has a choice between two methods of abatement. One method involves purchasing a significant amount of land on which to construct treatment ponds. The other method is to buy a set of mechanical devices. Under current tax laws, the investment in mechanical devices would be depreciable, while the investment in land would not. Thus, considering tax breaks, the mechanical approach might cost the firm less than the land-use approach, while the real cost to society, that is, the cost before taxes, of the land purchase method, would have been cheaper. (p. 1533)

These tax provisions may also influence abatement choices away from least-cost techniques which have high operating costs or which involve process changes if, as is often the case, the tax break applies only to facilities and equipment designed specifically for pollution abatement.

Another criticism of financial assistance based on tax concessions is that tax provisions are only of use to profitable firms and provide more aid to firms which pay higher corporate income taxes. Thus these programs provide the least support to those firms which require it most.

C. The Second-Best Externality Price

Despite the problems with externality pricing discussed in section III, many economists still argue that they compare favourably with the direct regulation-enforcement framework (Anderson et al., 1977; Baumol and Oates, 1988; Dewees, 1991).

A workable effluent charge is designed to achieve a predetermined environmental standard. A uniform charge per unit, that is, pound, ton, etc., of a particular type of pollutant, for example, SO_2, particulates, etc., is set for all major polluters to achieve some overall standard. Cost-minimizing firms will abate pollution in

response to such a charge as long as the cost of doing so, that is, the marginal cost of abatement, is less than the effluent charge. In essence, the individual firm abates pollution until the marginal cost of abatement, that is, the cost of removing an additional unit of pollution, equals the charge.[43]

While a uniform charge is set for each point source, a different charge would likely apply to each pollutant included in such a scheme. The overall standard must be determined by the environmental authority and should be stated as the amount of a particular pollutant which can be emitted into the environment over a certain period of time. For example, the overall standard might be such that all major SO_2 emitters in a jurisdiction would be allowed to emit one million tons of SO_2 into the air per year. This "standard," however, would not be translated into a restriction on the emissions of any particular point source. The method of achieving this standard would be to set an effluent charge and to alter this charge during each time period,[44] until the overall standard is achieved.

Under this scheme, the charge would have to be adjusted over time to maintain the overall standard in order to compensate for inflation and growth (Fisher, 1981). Continuing inflation would mean a decline in the real value of the charge, if its nominal value is fixed. This would result in emissions in excess of the desired level in particular years. Indexing the nominal charge to vary with the annual inflation rate would largely overcome the possibly costly necessity of annually altering the charge by administrative decree.

Growth of output over time and the resulting increase in the production of polluting by-products require increasingly stringent abatement if the overall standard is to be maintained (Magat, 1978). The incentive to increase levels of abatement may require increases in the real value of the charge.[45] Thus the charge might have to be adjusted upwards by administrative decree. The signal for such an adjustment would be given when emissions from the relevant point sources, in aggregate, exceed (or fall short of) the desired standard.[46] The charge would then be adjusted in order to achieve the overall standard.

This system must obviously be monitored, not to detect violations at individual sources but rather to determine the total emissions from each point source per year, that is, per time period, where the time period in question is the one over which the overall standard has been set. This would likely require taking a number of random samples of emissions from each point source which would then be used to calculate statistically the total emissions from that source during a particular time period.

These data would then serve two purposes:

(i) When aggregated for all relevant sources, to show whether the overall standard is being met and hence whether the charge should be altered;

(ii) To determine the payment by individual point sources which can be calculated as the charge, t, times the calculated emissions from the point source.

Monitoring some pollutants, such as SO_2, at certain sources, such as combustion point sources, may be somewhat simpler than for other sources. A knowledge of the sulfur content of the fuels being burned, the quantity of the fuels being burned, and the amount of sulfur being collected in abatement equipment would yield adequate estimates of emissions. Unfortunately such techniques cannot be applied to all pollutants or point sources. At any rate, continuous monitoring is not a requirement of this scheme.

This variant of the effluent-charge scheme compares favourably with the direct regulation-enforcement approach. Under an effluent charge there is no incentive to delay the installation of abatement equipment. While failure to install equipment does result in the same type of cost savings described earlier, it also results in a larger effluent-charge bill in this case. While there is no incentive for bargaining or delay of compliance once the charge has been set, there will likely be bargaining over whether this type of scheme should be imposed at all and what the initial level of the charge should be.

The effluent-charge scheme is a decentralized method of control. Once the charge is set, it is up to the firm to decide how to reduce pollution emissions. There are usually several possibilities: end-of-pipe treatment, process change, output reductions, input substitutions, change in product mix, etc. One advantage of such an approach is that since it is the firm which chooses the appropriate methodology, it will obviously do so in a manner which will minimize the costs of achieving a given level of abatement. Under the direct regulation-enforcement framework, in which a plant is sometimes ordered to adopt a particular abatement technology, the firm's flexibility of choice of abatement techniques is somewhat eroded. As a result, abatement costs at a point source will likely be higher under the direct regulation-enforcement framework.

A popular criticism of economic incentives, particularly effluent charges, is that they provide a "licence to pollute." Legislators and environmentalists often contend that when faced with a charge, firms will merely pay the charge and continue to pollute (Kelman, 1981). The empirical evidence, however, does not support this contention.

Much of the available evidence can be found in studies of a prototype of the effluent charge called "sewer surcharges." One study of poultry-processing plants located in various U.S. cities, Ethridge (1972), finds that for every 1% increase in surcharge rates, BOD[47] discharges per 1000 birds fell by .5%. Elliott and Seagraves (1972) in a study of the overall impacts of these sewer surcharges in the U.S. also find that industrial BOD emissions appear to respond negatively to the level of the surcharge. A 1% increase in the surcharge rate in a typical city was found to result in a .8% decline in industrial BOD emissions. Sims (1979) found considerable responsiveness to surcharge schemes by breweries in Canada: a 1% increase in the surcharge rate on BOD would lead to a .573% decline in BOD emissions from a typical brewery.

An analogue to the effluent-charge scheme is the beverage container deposit scheme in use in various North American jurisdictions. Consumers receive refunds when they return empty bottles or cans. This scheme acts like an effluent-charge on solid waste. Thus, persons who add these containers to the solid waste stream in essence pay a charge in terms of foregone revenues. Various studies suggest that even the rather small deposits charged under this scheme have been effective in limiting the can and bottle component of the solid waste stream. In Oregon, where a deposit of two cents is required on reuseable bottles and five cents on all other containers, the returns of refillable beer bottles increased from 31 percent to 96 percent of the market. In addition, bottle and can litter decreased by over 65 percent in the year after the introduction of this scheme. (Anderson et al., 1977).

The evidence provided by these studies of responsiveness to economic incentives and to analogous schemes such as the deposit system for cans and bottles, tends to refute the "licence to pollute" arguments.

With respect to cost effectiveness[48] and informational requirements, it has long been maintained that market-type schemes and especially effluent charges have advantage over the direct regulation-enforcement framework.[49] The argument compares the cost effectiveness of a simple effluent charge and of a uniform effluent standard in achieving a total reduction in the emissions of a particular pollutant. The effluent charge is adjusted until the required abatement is achieved. With the uniform standard, total abatement is divided equally among all of the point sources. Provided all point sources do not have identical control cost functions, the costs of achieving a given level of abatement will always be minimized with the charge.[50]

There are at least two problems with this argument:

(1) an effluent standard which is cost effective could obviously be designed;

(2) the cost effectiveness of these single-price market schemes disappears when firm location and pollution dispersion become important as they undoubtedly are in the real world.

The first problem implies correctly that almost any environmental policy can be cost effective, and as a result the relevant question involves the necessary informational requirements. The informational requirements for a cost-effective charge scheme are also affected by the second problem.

If the goal of the program is to limit the size of total emissions at least cost, the authority might set a standard limiting total emissions (by weight) to a certain amount (S). Assuming only two polluters, the optimization problem would then be:[51]

$$\min C^1(Z_1 - e_1) + C^2(Z_2 - e_2) - \lambda\,[S - e_1 - e_2] \quad (7)$$

where λ is the Lagrange multiplier.[52] The first order conditions suggest that:[53]

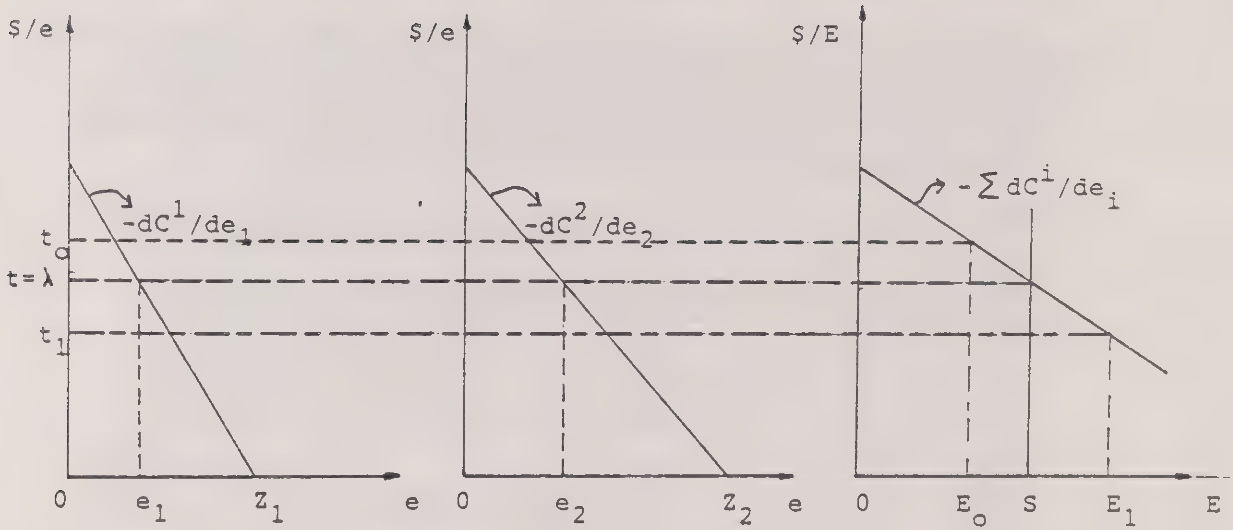
$$\frac{dC^1}{dA_1} = \frac{dC^2}{dA_2} = \lambda \qquad (8)$$

Thus each polluting firm should abate until its marginal costs of abatement are equal and the constraint $S = e_1 + e_2$ is met (see Figure V). Since we know that profit-maximizing, or cost-minimizing, firms will chose to abate until:

$$\frac{dC^i}{dA_i} = t$$

where t is the externality price, then it is clear that λ is the externality price which will achieve the aggregate standard at least cost.[54]

FIGURE V
COST EFFECTIVENESS OF EXTERNALITY PRICES



NOTE: $E = e_1 + e_2$

This pricing scheme has the advantage over an effluent standard scheme of needing less information. λ could be determined with little or no knowledge of abatement costs through iteration. Assume the authority sets an initial charge $t_0 > \lambda$. It would observe aggregate emissions $E_0 < S$, indicating that a reduction of t is in order. If it next sets a charge $t_1 < \lambda$, it will observe aggregate emissions $E_1 > S$, indicating that an increase in t is in order. This process would also provide cost information to the authority.[55]

On the other hand, an effluent standard approach would require knowledge on all abatement costs to be cost effective.[56]

Of course the iterative procedure presented above is not costless. If firms react to an initial charge such as t_0 by installing slow depreciating capital equipment, the reaction to changes in the charge might not result in a least-cost solution. Baumol and Oates (1988) suggest that advanced warning of the possibility of tax changes would allow polluters to build flexibility into their plant designs.[57]

While this externality price is not, in general, Pareto optimal (unless S happens to be the optimal aggregate standard), it does guarantee the given reduction of aggregate emissions at least cost. This appears to be the basis of one of the primary advantages of the externality pricing approach: its cost effectiveness.

The problem with the analysis to this point is the constraint chosen. Is it reasonable to set a standard based on the aggregate weight of the discharge? In some cases, where for example, the pollutant is conservative and the airshed is perfectly mixed, this may be a reasonable approach. Ultimately however, despite the difficulties of estimation, it is the damages of pollution emissions which are of concern. Thus a standard more closely aligned with damages seems desirable. If the pollutant is not conservative or if the environment is not perfectly mixed, an aggregate emissions standard is unlikely to have this property. Ambient standards at a receptor point would seem better able to fulfill this requirement.

Assume there is a single receptor and the target air quality, that is, concentration of pollutant, at that receptor is S_A. Also assume that a unit of emissions from point 1 increases the concentration at the receptor by α_1 and from source 2 by α_2. Thus a cost-effective policy would:

$$\min C^1(Z_1 - e_1) + C^2(Z_2 - e_2)$$

subject to:

$$\alpha_1 e_1 + \alpha_2 e_2 + B \leq S_A$$

where B is the background concentration of the particular pollutant which is unrelated to emissions from either polluter. (It is assumed that B is constant and less than S_A.)

The solution to the problem implies that:

$$\frac{dC^1}{dA_1} = \lambda\alpha_1$$

$$\frac{dC^2}{dA_2} = \lambda\alpha_2$$

Therefore, $$\frac{dC^2/dA_2}{dC^1/dA_1} = \frac{\alpha_2}{\alpha_1}$$

Thus even in this case an iterative procedure could be used. The effluent charges would differ between the two sources depending on the meteorological constants α_1 and α_2. However, the ratio of the two charges would be a constant equal to the ratio of the α's. Thus, the authority could set a charge of t on source 1 and $(\alpha_2/\alpha_1)t_1 = t_2$ on source 2. If the concentration of pollution is too high, t_1 should be increased. Since t_2 is defined in terms of t_1,it will follow suit. All that is required is that the authorities know α_1 and α_2.[58]

A more realistic model would incorporate multiple receptor points. Consider the case of two receptors where α_{ij} is the impact of a unit of emissions from source i on the concentration of a pollutant at receptor j. The least-cost solution can be derived by,

$$\min C^1(Z_1 - e_1) + C^2(Z_2 - e_2)$$

subject to:

$$\alpha_{11}e_1 + \alpha_{21}\,e_2 + B_1 \leq S_1$$

and $$\alpha_{12}e_1 + \alpha_{22}\,e_2 + B_2 \leq S_2$$

Assuming both constraints are binding, this would lead to the following conditions:[59]

$$\frac{dC^1}{dA_1} = \lambda_1\alpha_{11} + \lambda_2\alpha_{12} \qquad (9)$$

$$\frac{dC^2}{dA_2} = \lambda_1\alpha_{21} + \lambda_2\,\alpha_{22} \qquad (10)$$

Thus: $$\frac{t_2}{t_1} = \frac{dC^2/dA_2}{dC^1/dA_1} = \frac{\lambda_1\alpha_{21} + \lambda_2\,\alpha_{22}}{\lambda_1\alpha_{11} + \lambda_2\,\alpha_{12}}$$

Given that both constraints are assumed binding, the λ_i's are nonzero. The values of λ_1 and λ_2 can only be determined in a full solution to the above problem. Thus knowledge of the meteorological constants is no longer sufficient. The authority must also have knowledge of the abatement cost functions.

With n receptors, equation (9) becomes:

$$\frac{dC^1}{dA_1} = \sum_{j=1}^{n} \lambda_j \alpha_{1j}$$

It is usual that several of the λ_j's may be 0, that is, air quality at these receptors is better than the ambient standard when the standard is at least met at all n receptors. This result suggests that the trial and error approach to this problem would not be fruitful.

As well, the meteorological aspects of the problem are significant. The usual claim that prices are preferable to standards because they are cost effective must be qualified although there are certainly cost advantages to externality prices in some cases.[60] As the number of receptors in a non-perfectly mixed environment increases, the cost effectiveness of the externality price is harder to guarantee.

Several studies have attempted to determine the cost effectiveness of a uniform externality price in a non-perfectly mixed environment. Since the administrative costs of differential effluent charges which truly minimize abatement costs rise rapidly with the the number of different charges, it might be interesting to investigate the cost disadvantage involved with a uniform price.

D. The Cost Effectiveness of a Uniform Externality Price in a Non-perfectly Mixed Environment

Many simulation studies have been undertaken in the last two decades to determine the relative cost effectiveness of alternative policy instruments. These studies have at least two elements in common: the use of a dispersion model used to determine the α_{ij} parameters discussed in the previous section; and a set of abatement cost functions for the various sources located in the region under investigation.

The main issue is the relative cost effectiveness of direct regulation and the uniform externality price or, as it is referred to in the environmental pollution literature, the single effluent charge (SECH).

Atkinson and Lewis (1974) and Spofford et al. (1976) use similar models to analyze the impact of various policies on particulates and sulfur dioxide emissions in St. Louis and Philadelphia respectively.[61] The direct regulation scheme used

was designed to approximate State Implementation Plans (SIP) and involved a uniform percentage rollback of emissions at all sources in the model required to achieve the desired ambient standard at the receptor with the worst air quality. The studies also consider a single effluent charge (SECH), a zone effluent charge (ZECH) and the least-cost programming solution (LC).[62]

With respect to particulates, the results of the two studies are very similar, with the SECH achieving the standard at 9% to 15% of the abatement costs required under the SIP. However, under an effluent-charge scheme, the firm must pay not only the abatement costs but also the taxes on the continuing emissions.[63] When these taxes are included, the SECH achieves the standard at from 34% to 64% of the abatement costs required under the SIP.[64]

The sulfur dioxide (SO_2) results, however, present a different picture. The SECH abatement costs were 120% of SIP abatement costs and, when effluent-charge payments are included, this balloons to 239% of SIP costs.[65]

As pointed out in Bohm and Russell (1985) this apparent perverse result can be expected in situations in which high marginal-cost-of-control pollution sources have large impacts on receptor air quality.

A very simple model can be used to demonstrate this point. It is assumed that there are two sources of a particular pollutant located near a receptor at which air quality is of concern (see Figure VI(c)). Given the prevailing winds, only source 1 will affect air quality at receptor R. Thus a least-cost policy would impose control only at source 1.[66]

Further it is assumed that achieving the desired air quality at R requires a 50% reduction in emissions at source 1 from Z_1 to 1/2 $Z_1 = e_1^s$ in Figure VI(a). Under the SIP scheme, a uniform 50% reduction in emissions is imposed on all sources. Thus the cost of the SIP (C_{SIP}) is:

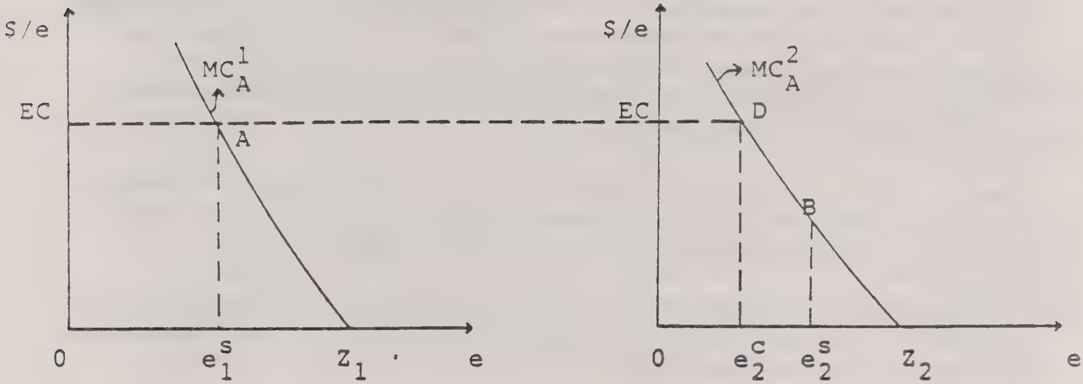
$$C_{SIP} = Z_2 e_2^S B + Z_1 e_s^1 A$$

$$\text{where } e_2^s = \frac{1}{2} Z_2 \text{ and } e_S^1 = \frac{1}{2} Z_1 .$$

The SECH required to reach the standard at R is EC, since this is the lowest charge that will induce source 1 to reduce emissions by the required 50%. The cost of the SECH (C_{SECH}) is:

$$C_{SECH} = Z_2 e_2^c D + Z_1 e_1^s A > C_{SIP}$$

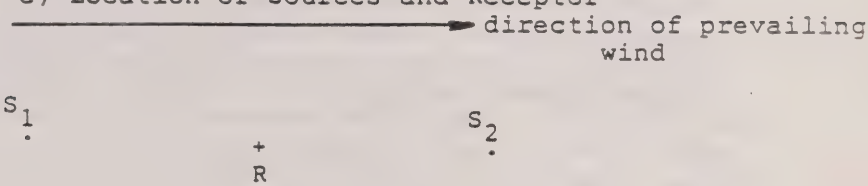
FIGURE VI
THE IMPACT OF LOCATION ON COST EFFECTIVENESS



a) High Marginal Abatement Cost Source (S_1)

b) Low Marginal Abatement Cost Source (S_2)

c) Location of Sources and Receptor



Note: S_1 is the location of the high control-cost source;

S_2 is the location of the low control-cost source; and,

R is the receptor at which air quality is of concern.

Thus even without taking the effluent charge payments[67] into account the SECH is more costly than the SIP rollback model.[68] The least-cost (LC) solution would involve a differential charge of EC at point source 1 and 0 at point source 2. Thus,

$$C_{LC} = Z_1 e_1^s A$$

It is thus clear that:

$$C_{LC} < C_{SIP} < C_{SECH}$$

However, if we look at the costs to the firm a rather unexpected result emerges.

$$FC_{SIP} = C_{SIP}$$
$$FC_{SECH} = C_{SECH} + EC(e_1^s + e_2^c)$$
$$FC_{LC} = C_{LC} + ECe_1^s$$

where FC_i is firm cost under policy i. It seems probable that:

$$FC_{LC} > FC_{SIP}$$

This will be true if $ECe_1^s > Z_2 e_2^s B$. Since the evidence from both of these studies is that effluent-charge payments are at least of the same order of magnitude as abatement costs[69] and since $Z_1 e_1^s A > Z_2 e_2^s B$, if source 1 is the high-cost source, the result follows.[70] Thus the firms will prefer the less efficient direct regulation mechanism to the least-cost scheme because of EC payments.[71]

Of these two results, the one which is of most concern is the first, that is, $C_{SECH} > C_{SIP}$. The question is how frequently is this likely to occur. As pointed out in Bohm and Russell (1985) this occurred in the Spofford et al. model because of the petroleum refineries with very high SO_2 abatement costs which were situated close to a critical receptor. The addition of low-cost home and commercial heating sources far from the critical receptor set up a situation very similar to our example. However, it might not be wise to impose an effluent-charge on these small dispersed sources.

Two more recent studies of NO_2 emissions, Seskin et al. (1983) and Krupnick (1986), find a similar phenomenon for Chicago and Baltimore, respectively. As in our example, abatement costs in both of these cases are greater under a SECH than under the direct regulation approach. Nevertheless all of these studies, along with several others reviewed in Tietenberg (1990), suggest large social cost advantages to economic incentive schemes[72] relative to the existing direct regulation approach.

There are, however, at least two flaws in these studies: one related to the abatement levels under alternative policies and the other related to the revenues generated by charge schemes.

Oates, Portney and McGartland (1989) point out a problem which they argue has resulted in overestimating the cost advantage of least-cost economic incentive schemes over direct regulation schemes. Their argument suggests that the disadvantages attributed to the SECH schemes by Spofford et al., Seskin et al., and Krupnick are likely overestimated.

They claim that in most situations, the high-cost scheme (whether it is the direct regulation or SECH scheme) generally achieves a higher ambient quality at non-critical receptors.[73] Since some maximum level of pollution concentration is usually set that must not be exceeded at any receptor, the attainment of the standard at a critical receptor, that is, that receptor at which air quality is worst in an uncontrolled state, usually implies a higher than required ambient air quality at some other receptors, especially under undifferentiating policies such as SIP or SECH. The problem with the studies conducted to date is that they assign a zero value to the additional abatement undertaken.

As Oates et al. (1989) point out, many of these studies confuse air quality standards with air quality levels. Just because two policies achieve the desired air quality standard does not mean they will achieve the same level. As was pointed out earlier, an undifferentiated policy achieving the same standard as the least-cost policy would actually achieve higher levels of air quality at receptors other than the critical one.[74]

When the benefits of the cleaner air at these other receptor points are included, Oates et al. find the benefits of the incentive scheme over direct regulation are reduced. It can also be argued that those studies which find cost disadvantages for the SECH scheme overestimate the disadvantage of this scheme.

Another weakness of several of the studies is their treatment of revenues raised by economic incentive schemes. At best, they see the revenues as neutral and, at worst, as an embarrassment. In the first case, it is recognized that the revenue from externality pricing does not involve any real resource cost. It is merely a transfer from polluter to government. Firms view this payment as unfair and unacceptable in a highly competitive world. However, as stated earlier, if the environment is truly a valuable and scarce asset, should not those who use it, to the exclusion of others, that is, polluters, not be charged for this use? Thus, it seems clear that an effluent charge is just and equitable. The different approaches depend on the interpretation of the ownership of property rights. Under current direct regulation schemes, firms are allowed to emit wastes free of charge up to some standard. They have in essence been awarded the property right to this part

of the environment. The alternative view is that the environment is an asset owned by society, and society can demand payments for the use of this valuable asset.

Ultimately, one must ask about the costs and benefits of collecting such revenues. The costs most often discussed in the economics literature are the transitional costs of layoffs and plant closings. These are commonly used as threats by some firms in response to the possibility of more severe environmental regulations. Yet there is little if any proof that current environmental regulations have done anything more than induce the earlier closing of economically weak plants. However, there is no guarantee that a more vigorous and costly externality pricing scheme will not have a more substantial impact. Dewees (1991) suggests that the judicious use of economic incentives might prevent such problems. For example, the imposition of a tax on Ontario Hydro that raised costs by five percent would reduce demand somewhat, but Hydro would not leave the province.

In a highly competitive international industry the impact might be somewhat more substantial. For example, if Canada followed a stricter environmental program than the rest of the world, Canadian industry would suffer a disadvantage in the short term. In the long run, however, we would benefit from a cleaner environment and firms which are more efficient and knowledgeable about pollution control.[75]

There are also some benefits from the collection of revenue generated by externality pricing. First, because firms are required to pay for all of their pollution emissions, the price of their products is more likely to reflect the true social costs of production. Indeed, one might even argue that a plant closing brought on by an appropriate externality price is desirable, since it implies the firm's final output has a social value less than that of the resources (including the environmental quality) used in its production. In the absence of such charges, it may be argued that the firm is using an input, the environment, at an artificially low price and is thus producing too much output from society's point of view.

Second, externality price revenues can be used to reduce non-environmental resource misallocations. The externality price, or tax, is unlike other taxes: a regular tax, levied to raise revenue, has the side effect of distorting behaviour with the result of a resource misallocation. The externality price, on the other hand, is levied to change behaviour, that is, distort it towards more acceptable norms, and has the side effect of raising revenue. Thus, unlike conventional taxes, these are not accompanied by a dead-weight loss.

As is pointed out elsewhere (Oates, 1988) these tax rates should be based on environmental objectives, not revenue rationales. Nevertheless, revenues raised from these externality prices could replace existing, distorting taxes and would thus reduce the misallocation inherent in the tax system. This means that charge revenues should be diverted to the public treasury.[76]

Estimates of the magnitude of the reduction in this misallocation, for a hypothetical U.S. effluent charge on sulfur oxides and particulates, are made by Terkla (1984). He estimates revenues of from $1.8 to $8.7 billion in 1982 U.S. dollars. From available estimates, he assumes a marginal welfare loss of $.35 from each $1 of income tax revenue and $.56 from each $1 of corporate income tax revenue. He concludes that the efficiency value of revenue raised from these externality prices would be between $630 million and $3.05 billion if it replaced the labour income tax and between $1 billion and $4.87 billion if it were substituted for the corporate income tax.

A more recent study by Ballard, Shoven and Whalley (1985) suggests marginal excess burdens of capital taxes at the industry level as high as .463 and from income taxes of .314. As they point out, these estimates are sensitive to marginal tax rates. Since these estimates are for the U.S., and since Canadian marginal tax rates are somewhat higher, the gains from this approach might be even greater in Canada.[77]

It is clear that none of these simulation studies have considered the advantages offered by the "efficiency value" of revenues from externality pricing. Most present this revenue in a negative light, as a political liability. There is little doubt that there is corporate opposition to effluent charges and other forms of externality prices and is, at least to some degree, a result of opposition to the collection of this new tax. But if this tax were used to reduce an existing, distorting tax such as the corporate income tax, this opposition might be tempered to some degree. However, the purpose of the externality tax is not to generate revenue. This is but an agreeable side effect. Indeed, if the externality price is successful, revenues will fall over time.[78]

In conclusion, the large majority of simulation studies do find significant cost advantages to economic incentive schemes over the direct regulation approach. A further advantage of externality pricing is the "efficiency value" of the revenues generated. Given the advantages of economic incentive schemes it is surprising that they have not become more popular. Externality pricing is probably most prevalent in Europe, but even there it is recognized that these schemes are generally not designed to provide incentives to abate, but rather are designed to raise revenues which are used to administer environmental programs and to subsidize pollution investments (OECD, 1989). The solution to this conundrum, it appears, must be sought elsewhere.

E. Dynamic Efficiency

The previous section suggests that the cost effectiveness or static efficiency of externality pricing may be somewhat suspect under certain circumstances. However, one of the strongest arguments for an externality price is that it is more inclined to induce technological change in abatement techniques than is direct

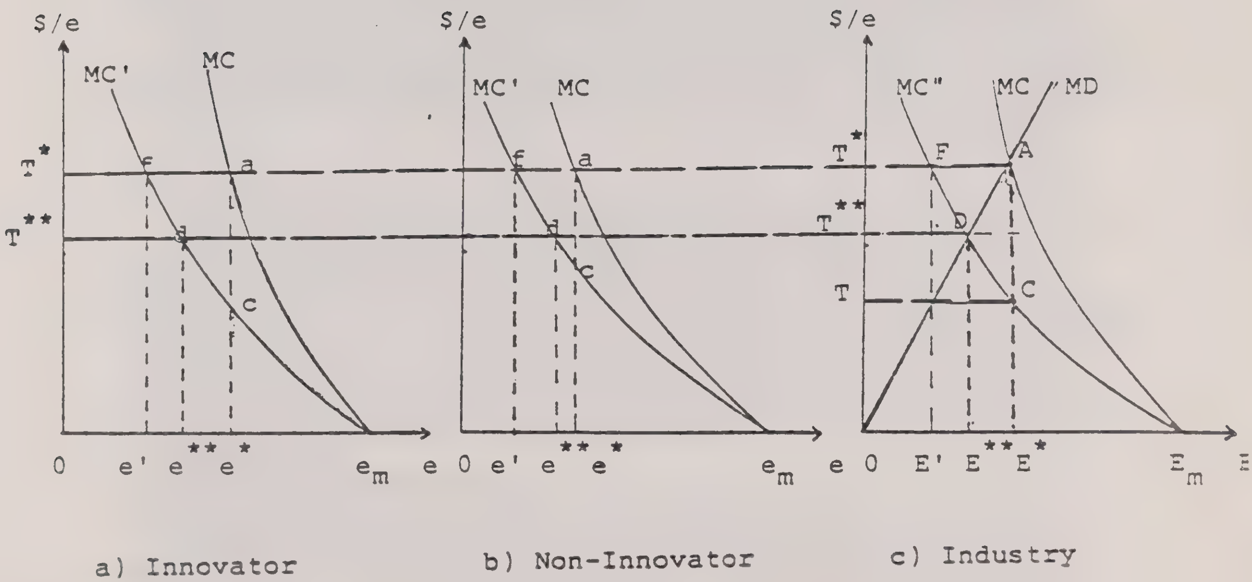
regulation. It is thus more efficient in a dynamic sense. For example, Dewees (1980) argues that under a standard scheme, there is no incentive to develop a technology that achieves more abatement than is required by the standard. Effluent charges however, provide a continuous incentive to abate. If so, then abatement costs will decline more quickly over time under an externality price and in the long run may be more important than static efficiency, that is, cost effectiveness.[79]

Milliman and Prince (1989) present a more complete comparison of the dynamic efficiency of effluent charges and the direct regulation approach.[80] As they point out, most analyses of technological change are firm specific. They look at how a technological change affects a firm's costs (Dewees, 1980; Bohm and Russell, 1985). However, as Milliman and Prince point out, costs are reduced not only by these firm-specific innovations but also by diffusion of the innovation to other firms and an optimal agency response to these changes. They argue that the cost reductions brought about by these latter two changes likely dwarf those of the firm-specific gain. As well, any delay in these changes will reduce the overall gains from an innovation.

Their comparison of alternative policies is based on a model presented in Figure VII. Assume that there are N competitive firms emitting a homogeneous waste into a perfectly mixed environment. Initially the optimal level of emissions is E^*, which may be achieved with a charge of T^* or an effluent standard of e^* against each firm, where $E^* = Ne^*$. Assume that a single firm in diagram (a) develops an abatement innovation which shifts the marginal cost of control from MC to MC′. This does not initially affect the industry marginal abatement cost (MC in diagram (c)) since the innovating firm is small relative to the market.[81]

Before the innovation, the pollution-related costs of the innovating firm include abatement costs, $e_m ae^*$, and tax payments, e^*OT^*a. As a result of the innovation, the firm in diagram (a) reduces its emissions to e′ and finds that its pollution-related costs include abatement costs $e_m fe'$ and tax payments of $e'0T^*f$. Thus the firm's gain from the innovation under a charge scheme is $e_m fa$. Under a direct control scheme, emissions remain at e^* and the gain to the firm is thus $e_m ca$. The incentive for the firm-specific innovation is obviously greater under the charge scheme.

FIGURE VII
DYNAMIC EFFICIENCY WITH STANDARDS AND CHARGES



Widespread diffusion of the innovation in the industry will eventually cause the MC curve of the (N - 1) non-innovating firms to shift to MC′ and the industry marginal cost of control thus shifts from MC to MC″.

It is assumed that the agency does not initially perceive this change and thus does not alter its policy. Since there is no royalty system there is no further gain to the innovating firm under either of the schemes. Indeed, diffusion may threaten the innovator's gains through increased product market competition.

There is certainly a gain to the non-innovating firms. But once again the incentive to promote the innovation (by other firms, given that diffusion is expected) is greater under the charge scheme. Under the charge scheme the non-innovative firms in diagram (b) save $e_m fa$; whereas under direct controls each one saves $e_m ca$.

Finally, it is assumed that the authority adjusts the aggregate emissions to the optimal level of emissions E^{**} and each firm to e^{**} where, $E^{**} = Ne^{**}$. Under the charge scheme, T^{*} is lowered to T^{**}.

Under the charge scheme, emissions decline to E^{**}. The industry gain from this optimal alteration in the charge is shown in diagram (c) as $DT^{**}T^{*}F$. The principal concern here is the industry gain since it indicates the propensity for political action, that is, the propensity of the industry to demand an optimal reaction by the government.

With direct controls, optimal behaviour by the authorities results in aggregate emissions E^{**} and firm level of emissions of e^{**}. This results in a loss of $E^{*}E^{**}DC$. Thus under direct controls there is no particular reason to expect industry pressure for optimal adjustments.

The overall gain from the innovation, diffusion and optimal policy adjustment can be summarized as:

a) under charges: $E_m FA + T^{**}T^{*}FD > 0$

b) under direct controls $E_m CA - E^{*}E^{**}DC \lesseqgtr 0$

This analysis suggests that there is not much of an incentive to innovation under direct regulation, especially if the government adjusts standards in an optimal manner to perceived declines in cost. Obviously the externality price is much more likely to induce technical change. Indeed if it is perceived that there are potentially substantial gains from the entire process, it is more likely that the initial innovation will take place.

Figure VII could also be used to investigate the dynamic efficiency of a second-best standard and charge. Assume the authority sets the standard E^{*} and

changes the charge iteratively to retain this level of emissions. With respect to the standard scheme, the authority sets an aggregate standard of E^* which translates into firm standards of e^*. These are not adjusted.[82]

The earlier conclusions still hold regarding the relative cost savings under the two schemes (charges and direct controls) from the firm-specific innovation and diffusion of the innovation. The third stage in the process is, however, altered. Since the authority does not alter E^*, there is no loss to firms under the standard scheme. With the charge scheme however, the authority will adjust the charge downward to T in order to retain aggregate emissions E^*. The industry gain from this adjustment is even greater than under the optimal adjustment analyzed earlier. It is TT^*FC.

The overall gain from the innovation, diffusion and second-best policy adjustment rule can be summarized as:

a) under charges: $E_mCA + TT^*AC > 0$

b) under direct controls: $E_mCA > 0$

Under this second-best policy rule, there is a definite incentive for innovation under direct regulation[83] but it is certainly smaller than under the externality price scheme.[84]

F. Enforcement and Monitoring

Most of the analysis carried out to this point is characteristic of the literature in this area, in that it is based on the assumptions that monitoring and enforcement are costless and that firms voluntarily comply with environmental controls. Monitoring and enforcement costs however, do have a definite impact on the analysis of optimal externality prices[85] (Lee, 1984; Harford, 1978) and, it is claimed, the relative desirability of externality prices. On one hand, a common claim is that these monitoring costs are a particular disadvantage to pricing schemes but it is also sometimes claimed that charge schemes have enforcement advantages over direct controls (Russell, Harrington and Vaughan, 1986).

When individuals or firms are required against their self-interest, it seems unreasonable to expect voluntary compliance; hence the use of penalties or sanctions. However, since the authorities must first identify those sources not in compliance, monitoring is a necessary prerequisite to effective environmental control.

The nature of emissions and the behaviour of polluters make monitoring difficult. Since pollution sources are hard to identify after the fact, monitoring must take

place at the source. As well, counter to a common assumption, discharge sources do not have perfect control over emissions. Operating changes, human error, equipment failures, all affect discharges. As a result, a polluter who installs the appropriate control equipment and operates it effectively may still be observed in violation a small percentage of the time (Russell, Harrington and Vaughan, 1986). On the other hand, polluters may not comply, especially when penalties and monitoring are ineffective. Differentiating between the compliant polluter whose random emissions are observed in excess of the standard at a specific time and the non compliant polluter is likely impossible. Thus some innocent firms may be convicted of violating standards.[86]

This stochastic nature of emissions when combined with the imperfection of monitoring equipment introduces some ambiguity to the notion of enforcement. This problem is compounded by uncertainties regarding the legality of entry to plants for the purpose of monitoring. It is often contended that for safety reasons the monitoring agency must announce intentions of a visit in advance, a requirement that seems likely to introduce some bias into the data collected.

Monitoring certainly creates problems for environmental programs, but why should these problems be any more severe for charges than for standards? It is sometimes alleged that design or technological standards are easier to monitor, since instead of requiring a certain level of abatement or emissions, they require a certain type of equipment. Such an approach in the U.S. involves the observation of installation and testing of the required equipment. (Russell, Harrington and Vaughan, 1986). A similar monitoring procedure is used for automobiles.

Indeed this type of monitoring may offer fewer logistic or cost problems,[87] but ultimately, it is ineffective. It provides no incentive for the polluter to maintain or even operate the machinery in an effective manner. Monitoring of the emission stream is required if continuous compliance is to be assured.

Dewees (1980) suggests that all environmental programs require the accurate measurement of emissions. He argues that opposition to effluent charges on the basis of monitoring difficulties is actually an objection to vigorous enforcement.

An appropriate comparison of the monitoring and enforcement costs associated with alternative environmental programs should begin with the assumption that both schemes are being applied in a vigorous and effective manner. Dewees, Everson and Sims (1977) suggest that given such an assumption, effluent charges may require less monitoring than a system of direct controls. In the latter case, the environmental authority monitors in order to catch firms in violations. A firm in violation is prosecuted and a fine levied, preferably an amount related to the damage created by the violation. But to ensure that the polluter faces an expected fine which approximates damages, monitoring would have to be very stringent.[88]

Alternatively, with an effluent charge it is only necessary to determine the average emission rate. This can generally be done with a reasonable degree of confidence with a finite sample size. In the latter case the frequency of monitoring is designed to increase the precision of the estimates, whereas in the former, it is actually an enforcement tool designed to encourage compliance. It appears that monitoring frequency under the charge scheme would be no greater than under an effective performance standard.

It has been suggested that the cost of monitoring should be imposed on polluters through self-reporting (Anderson et al., 1977), an approach compared to the income tax system. There is however, a non-symmetry here since with income taxes there are records of transactions which can be checked by authorities of the tax department, whereas it may be far more difficult to confirm a measurement of pollution long after the emission has taken place.

Recent work on the issue of differential enforcement effort under charges and standards using self-reporting (Harford, 1978; Linder and McBride, 1984) suggests that charges might be more difficult to levy than enforced compliance to standards when both are imperfectly enforceable. A simple model from Harford can be used to demonstrate this result.

A profit-maximizing firm produces an output x. Its revenues are R(x) and its costs C(x,e) where e is actual emissions. The firm reports its waste emissions to the environmental authority, e_R and pays a penalty for evasion f(v) where v, the violation, is $e - e_R$. The probability of being caught in violation is p(v) and thus the expected fine for under-reporting is,

$$G(v) = p(v)f(v).$$

The firm is assumed to maximize profits (π_t) where,

$$\pi_t = R(x) - C(x,e) - te_R - G(v) - pvt$$

If the polluting firm is caught under-reporting, it faces an expected fine of G(v). It must also pay the charge on the under-reported emissions (i.e. vt). The first-order conditions for a maximum are:

$$R_x - C_x = 0 \tag{11}$$

$$- t + G' + pt + p'vt = 0 \tag{12}$$

$$- C_e - G' - pt - p'vt = 0 \tag{13}$$

Rearranging (12) and (13):

$$t = G' + (p + p'v)t \qquad (12')$$

$$- C_e = G' + (p + p'v)t \qquad (13')$$

Equation (12′) means that the firm equates t, the cost of a marginal increase in e_R to $G' + (p + p'v)t$, the reduction in the total expected penalty for a marginal decrease in v. Equation (13′) states that a profit-maximizing firm should equate the change in control costs from a marginal increase in emissions to the increase in expected penalties from a marginal increase in emissions.

Combining (12′) and (13′) implies:

$$- C_e = t$$

The firm reduces e until marginal control costs equal the tax. Thus the true emission level of the firm does not depend on the size of the fine or probability of being caught for under-reporting. On the other hand, equation (12′) implies that an increase in t will cause the marginal expected fine to rise. Since (from the second-order conditions) the slope of the expected fine function with respect to v is positive, this implies an increase in v. Indeed, using comparative statics, Harford shows that:

$$\frac{\partial e}{\partial t} < 0; \quad \frac{\partial e_R}{\partial t} < 0 \text{ and } \frac{\partial v}{\partial t} > 0$$

Thus, an increase in t designed to reduce emissions increases the size of the violation which a firm will choose and results in a more difficult enforcement problem. Harford (1978) suggests that this is an asymmetry between charges and standards.

In a more recent paper, Harford (1987) suggests that this finding of an asymmetry was in error. In the earlier work, the fine under the standard scheme was levied for violations of the standard. If there was self-reporting it was assumed to be done honestly. This is the source of the asymmetry. In the 1987 paper, Harford notes that if self-reporting is required under the standard a similar kind of behaviour will be observed. He notes ". . . a pollution tax is essentially a fine for violating a standard that allows zero pollution." (pp. 293-94) Thus he finds with an imperfectly enforceable standard that ". . . [g]reater rates of fines on reported violations of the standard, while reducing actual pollution, will tend to increase the amount of under-reporting . . ." (p 301).

If there is an asymmetry with regard to the monitoring required for charge and standard schemes, it appears to be a result of an asymmetric approach to the rigour with which the schemes are applied. When both schemes are designed to

achieve a pre-designated ambient standard, there is little evidence that requirements for charges are more excessive than those for direct controls. On the other hand, direct control schemes which are designed to placate environmental groups, with little hope of achieving significant environmental improvements will most definitely have an edge in terms of both monitoring and enforcement costs.

G. Distributional Considerations

Various studies maintain that the impact of environmental policy on the distribution of income is regressive (Asch and Seneca, 1978; Baumol and Oates, 1979). It is usually argued that environmental quality is a luxury, and thus has a relatively high income elasticity of demand. As well, the poor are less able to protect themselves from the dislocations which sometimes accompany environmental policies.

It is difficult to deal with the first argument. The rich are willing to spend more of each dollar earned on environmental concerns. Thus any environmental improvement may yield greater gains to the rich. Any serious environmental policy which achieves a cleaner environment may benefit the rich more than the poor but this does not appear to be a basis for favouring one policy over another.[89]

The second rationale deserves more consideration. It is tempting to argue that because the financial impact of an appropriate externality price may be large, this policy will create greater dislocations than direct controls.[90] Indeed, such arguments have been used in favour of the direct regulation approach to environmental policies.

Adopting an environmental policy because of its impact on the distribution of income however, seems incomprehensible. Since the purpose of these policies is to improve environmental quality, not to achieve an equitable distribution of income, it seems reasonable to leave the latter to the fiscal authorities.

Quinn (1983) believes, however, that it seems unlikely that current environmental policy makers have adopted policies designed to advance distributional goals of the type discussed above, although he believes that the distribution of wealth is a significant factor in the choice of direct controls over externality pricing.[91] His argument is based on two premises:

(1) The political process will always attempt to maintain the status quo distribution of wealth.

(2) A redistribution of wealth using fiscal policies is politically difficult.

Premise (2) rules out the adoption of a policy which alters the status quo distribution of wealth, since there is virtually no acceptable way to compensate the losers.[92] Based on this theory, the current status-quo division of property rights involves an allocation of the environment, up to current standards and maybe beyond, to polluters. A program of externality pricing would drastically change this distribution and hence would, one might argue, be politically unacceptable.

This status-quo distribution of property rights cannot, it appears, be justified on equity[93] or efficiency grounds. Such political arguments thus appear to provide significant insights into the choice of environmental policies.

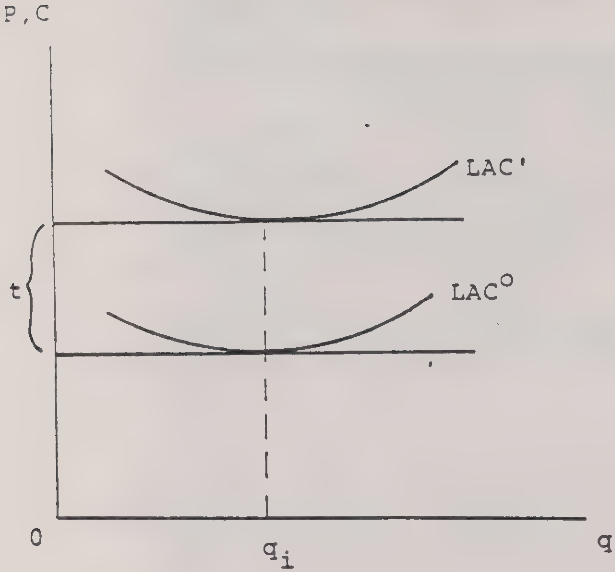
Burrows (1980) also provides support for this contention. He states that ". . . the implications of say a change in property rights for the general distribution of income in society may be less important than the particular income (and other welfare) consequences for the gainers and losers from the change. In fact, the legal system or pollution control agency may studiously ignore the general implications for the distribution of income, in an effort to provide protection for losers that is not dependent on income levels of parties involved in the conflict of interest." (pp. 53-54)

Buchanan and Tullock (1975) provide further enlightenment on the choice of standards over charges. They use the competitive industry model to determine polluter profits under a charge and under a standard scheme. Assume there are n identical firms in a constant-cost, perfectly competitive industry. The marginal damage of pollution is assumed to be constant per unit of output. Initially each firm produces Oq_i as shown in Figure VIII(a), with long-run average cost, LAC°, price, P°, short-run supply, SRSS° and long-run supply, LRSS°. Industry output is

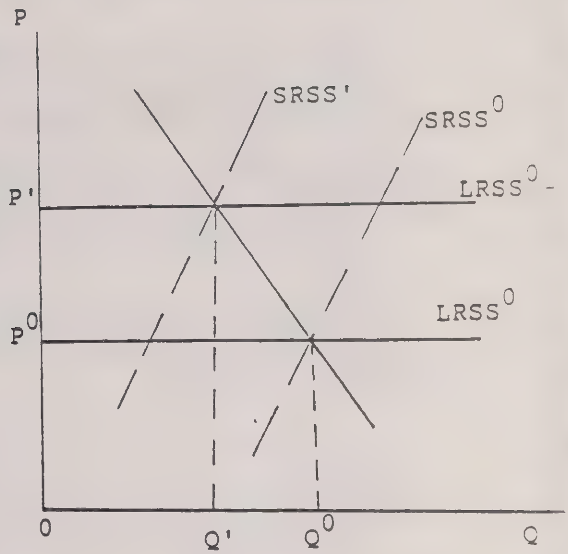
initially $Q^\circ = \sum_{i}^{n} q_i$.

Now assume the environmental authority levies a tax, t, equal to the marginal damage per unit of output.[94] This shifts the LAC vertically upward by t at each output level and, initially shifts SRSS° up by the same amount. In the short run, this will result in losses and firms going out of business. As firms exit, the SRSS shifts further left until, in the new long-run equilibrium, it attains the position SRSS′. The comparative static results show that firm output remains constant at q_i, the number of firms in the industry declines to $n' < n$, industry output falls to Q′ and market price of output rises by the amount of the tax to P′.

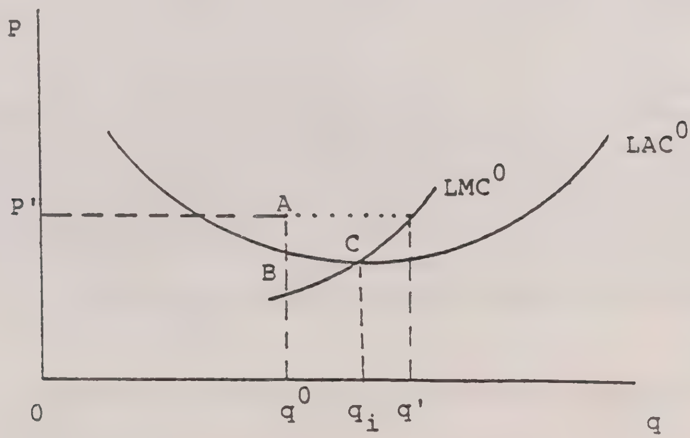
FIGURE VIII
DISTRIBUTIONAL IMPACT OF STANDARDS AND CHARGES



a) Firm Equilibrium (Charge)



b) Industry Equilibrium (Standard)



c) Firm Equilibrium (Standard)

A similar result can be attained with direct controls. The authority first determines the optimal industry output of Q' and divides it between the n original firms. Thus firm output is $q^\circ = Q'/n < q_i$. Thus under this standard each firm decreases output by 100 $(Q^\circ - Q')/Q^\circ$%. Since industry output becomes Q', market price becomes P' (based on the market demand in (b)). Firms find themselves at a point such as A in diagram (c). At this position they are earning above-normal profits and hence there is an incentive for entry. As well, firms would like to expand output since at q°, $p' > LMC^\circ$.[95]

The policing task is much different from that needed under the charge scheme. The authorities must ensure that firms do not produce in excess of the quota (q°) and they must also prevent new entry, otherwise the optimal level of externality (defined by Q') will be exceeded. Of course, on efficiency grounds the tax is preferred to the quota since overall the cost of producing Q' under the standard is $(q^\circ B) \times Q'$ whereas with the charge (ignoring charge revenue since it is a transfer) the cost is $(q_i C) \times Q'$ which, as can be seen in diagram (c), is lower.

It is clear that polluters will prefer the standard scheme to charges. Under the tax, firms make short run losses and regain a normal return only after the exit of some firms, whereas with the standard there may even be a pecuniary gain. Indeed the quota system described is equivalent to a cartel. Despite each firm's motivation to cheat, it remains in their interest to seek regulations which enforce the quotas.[96]

But this does not explain the popularity of quotas or standards over the more efficient alternative of charges. Buchanan and Tullock argue that the distributional effects of charges and standards are the explanation. The tax alternative yields revenues which may benefit a large heterogeneous group, whereas the quotas benefit a small, concentrated, identifiable group. Because of its size and wealth, the latter group likely has more political power, and its probable gains per decision-making unit are much greater.

As Buchanan and Tullock (1975) point out, "The penalty tax amounts to a legislated change in property rights, and as such it will be viewed as confiscatory by owners and employees in the affected industry." (p. 142)[97]

This group will oppose effluent charges and use its considerable political power to push for a quota system. Dewees (1983) argues that, given the necessity of environmental regulation, labour and existing firms will prefer differentiated direct controls, involving stricter standards for new sources, to any of the economic incentive schemes that are currently available.[98]

Buchanan and Tullock (1975) conclude, "If the economist ties his recommendation for the penalty tax to an accompanying return of tax revenues to those in the industry who suffer potential capital losses, he might be more successful than he has been in proposing unilateral or one sided application of policy norms." (p. 143)

Dewees (1983) concurs and concludes, "If the political process responds to interest groups that are strongly affected by a policy, the effluent charge and sale of effluent rights are in deep political trouble unless the revenues are used to compensate the losers." (p. 70)

The chief advantage of a direct regulation policy appears to be based on the distribution of wealth. Apparently it is not a concern with the inequity of the distribution of incomes but rather a concern with the maintenance of the status-quo distribution of wealth that has led to the adoption of direct controls. Far from being merely an irrelevant curiosity, this point signifies that if externality prices are to be adopted they must somehow be transformed into a more politically acceptable form less threatening to the status-quo distribution of wealth.

The real-world externality pricing schemes conform well to this analysis. Most European schemes are prototypes of the German scheme reviewed by Brown and Johnson (1984). This effluent-charge scheme is an appendage to the existing direct regulation approach.[99]

Each source of water pollution[100] is issued a permit which specifies standards[101] which vary by industry and location. This permit also contains the information necessary to calculate the firm's charge payment, such as the expected concentration and volume of emissions. The charges levied in this scheme are based on the expected rather than the actual level of discharges.

Dischargers in compliance with minimum standards have the charge liability halved. If actual discharges (using the average of the last five observations) exceed these standards, the polluter faces legal actions as well as the loss of the 50 percent reduction in the charge obligation.

The revenue from this charge is used for water quality management, administrative expenses and for subsidies of waste treatment investment. Significant investment subsidies are available in most jurisdictions.

This charge scheme, like those in other European jurisdictions, for example, Italy and the Netherlands, is primarily a revenue-raising device which is an appendage to a policy of direct regulation. For many such schemes, the charge is too low to induce compliance with standards (OECD, 1989).

The German effluent-charge scheme's main effect comes from the discount available to those who achieve the minimum standard. This, along with fines for exceeding the standard, provide the main incentives for compliance. The incentive effect of the charge itself is questionable since average control costs appear to be about four times the charge (OECD, 1989).[102]

The primary incentive effects of the German scheme are similar to those from another form of economic incentive: the noncompliance penalty. In Connecticut, the Department of Environmental Protection has adopted a financial noncompliance penalty to supplement point source effluent standards. In essence, the authorities decide that a particular type of abatement equipment should be in place at a particular point source in order to achieve the required standard. A compliance order,[103] along with a deadline for compliance is then issued. If the equipment is not installed by the deadline, a noncompliance penalty, equal to the firm's cost savings may be imposed. The cost savings, including any profits made on this money, are calculated using a capital budgeting formula. In theory, this penalty makes compliance very attractive because compliance, that is, investment in the required abatement equipment, yields as good a return, in terms of foregone assessments, as do alternative investments.

The steps to calculate the penalty are:

(1) calculation of the relevant control technology for a given source;

(2) determination of installation, operating and maintenance costs for that technology from various engineering studies; and

(3) translation of this cost information into a measure of the cost savings of noncompliance per time period, for example, month.

Under this scheme, the Connecticut Department of Environmental Protection has the power to impose civil assessments without waiting for court action.

It is clear that the real-world effluent charge schemes[104] deviate sharply from the textbook welfare-maximizing Pigouvian tax discussed in section III, as well as the second-best cost-effective externality price discussed in section IV(c). The most effective real-world externality prices are noncompliance penalties. They are designed as appendages to direct control policies and have few of the static and dynamic efficiency benefits discussed earlier in this report.

V. Externality Pricing in Passenger Transportation

A. Introduction

The ultimate purpose of this study is to investigate the potential applicability of externality pricing to air pollution and noise externality problems inherent in passenger transportation. Potential and actual schemes are evaluated based on criteria inherent in the previous discussion in this report.

With respect to air pollution, transportation sources in North America contribute in excess of 47% of nitrogen oxide (NO_x) emissions, 71% of carbon monoxide (CO) emissions and 39% of hydrocarbon (HC) emissions (Button, 1990). As well, currently in developed economies, motor vehicles contribute about 40% of total carbon dioxide (CO_2) emissions (Button, 1990). Given the magnitude of transportation's contribution to environmental problems, the investigation of alternative control policies applicable to this sector is worthwhile especially since current government regulatory programs may be adding to, rather than reducing, the problem (Button, 1990).[105]

The automobile is the primary source of air pollution in passenger transportation and, as such, dominates discussions of environmental externalities in the academic literature. There is virtually no discussion of pollution problems from air transportation, public transit or rail. Indeed, these are very often considered to be part of the solution to the problem of auto pollution: diversion of passengers from automobiles to public transit would reduce pollution because there is a lower rate of emissions per passenger-mile from public transit. The relevance of such solutions will be discussed in the next section.

Congestion is an issue often related to air pollution. While this study does not directly address the issue, it should be emphasized that optimal congestion policy may, in some situations, conflict with optimal noise and air pollution policy. As Button (1990) points out ". . . measures to spread traffic more evenly across an urban area or over time, may well reduce congestion but equally may divert significant traffic flows into quiet residential areas, or encourage trips at otherwise peaceful times of the day, and thus have adverse environmental effects. Environmental externalities therefore require a specific policy response." (p. 62)

As with air pollution, the literature on noise pollution, is also dominated by a single mode, that of noise pollution at airports from jet aircraft. Part of the reason for this emphasis is practicality: the impossibility of separating the noise of other forms of transportation, for example, trucks and cars, from background noise, the large numbers of such vehicles, and the uncertainty of their path and time of travel present a virtually intractable monitoring problem.[106] But it is possible to monitor noise pollution with aircraft (Baumol and Oates, 1979) partly because information is readily available on flight times and paths.

Externality prices, as summarized in the first section of this report, may take at least three forms:

(1) they can be used to optimize or maximize the social surplus;

(2) they can be designed to achieve a predetermined standard at least cost; and

(3) they can be used to induce compliance with given standards.

(1) is the Pigouvian tax, (2) is the cost-effectiveness or standards-and-charges approach and (3) is basically a noncompliance penalty. From society's point of view, the net benefits of these forms of tax decline from (1) to (3), but so do informational requirements. Thus the ranking of these policies when such informational requirements are considered may change depending on the specific situation.

The last two sections of this report look at the possible gains or losses from variants or combinations of these forms of externality pricing, first for automobile air pollution and then for airport noise pollution.

B. Air Pollution

i) Introduction

The efficacy of current policies to control air pollution from transportation sources, that is, from the automobile, has been widely questioned in the economics literature (see Dewees, 1974; Mills and White, 1978; Crandall et al., 1986; Tietenberg, 1988). The major concerns are the lack of flexibility of regulations across regions and the lack of incentive for technological change. A first-best Pigouvian tax cannot be imposed on auto pollution emissions because of uncertainty surrounding the damages of pollutants from automobiles, but some form of second-best externality price might improve upon the current situation.

As was suggested in section IV(B) there are significant flaws in the direct regulation approach. Its inherent incentive to delay compliance has obviously characterized the North American auto industry for at least the past two decades. Of course, past problems with the program do not mean that it is failing today. After all, hydrocarbon (HC) emissions, carbon monoxide (CO) emissions and nitrogen oxide (NO_x) emissions have been projected to decrease relative to their 1970 levels by 20%, 40% and 78% respectively, despite an estimated 58% increase in vehicle-miles traveled (Crandall et al., 1986).

To determine whether an alternative approach would yield significant advantages, one must look at the current and projected problems of the program. This section

begins with an investigation of problems inherent in the current regulatory approach.[107] Some alternative schemes suggested in the economics literature are presented and evaluated. Finally, a survey of economic incentive policies actually used to alleviate passenger transportation air pollution problems is provided.

ii) Performance Problems with the Direct Regulation Approach

Crandall et al. (1986) list several defects of the current performance standard approach used in Canada and the U.S.:

1) Current regulation imposes more stringent standards for new cars, which prolongs the life of used cars by reducing the incentive to scrap them.[108] Since older cars are by definition "dirtier" cars, this may cause adverse short-term effects on emission levels (Gruenspecht, 1982). Thus the form of regulations on new sources used to regulate auto pollution depends for its success on changes in the composition of the fleet; an increased scrapping rate is crucial to its success.

2) The punishment inherent in current regulations is too severe. Preventing a major manufacturer from marketing a line of cars is not a credible threat. As a result the history of environmental regulation in the auto industry has been one of delays and postponements.

3) New source performance standards provide no incentive for owners to maintain the emission devices on their automobiles. While manufacturers are required to provide five-year 50,000 mile warranties on this equipment, there is little incentive for drivers to exercise this option since they are not liable[109] for poor performance. The problem is that the current legislation is focused primarily on the initial certification of the car, not its in-use emission rate.

These are the apparent structural flaws, but what of the scheme's performance in practice? The ultimate purpose of the program was to reduce suspected severe health risks from photochemical oxidants (smog), formed as a result of a complicated chemical reaction between HC, NO_x and sunlight.[110] However, since it is difficult to evaluate a program directly on that basis, it is more reasonable to look at aggregate emission rates and ambient air quality.

The EPA has produced models to predict aggregate emissions from automobiles over time. These emissions depend not only on new source standards but also on other variables such as the composition of the fleet by model and vintage, total

miles driven and distribution of mileage by model and vintage. A simulation of this model led to the following results (Crandall et al., 1986):

Year	Emissions (teragrams) HC	CO	NO_x	Vehicle-Miles Traveled (billions of miles)
1970	8.53	47.61	3.89	910.0
1975	6.52	39.39	4.37	1050.5
1980	4.44	29.21	3.67	1129.8
1982	3.94	26.37	3.50	1191.6

Between 1970 and 1982, despite a 31% increase in vehicle-miles travelled, emissions of HC, CO and NO_x fell by 54%, 45% and 10%, respectively. As the phase-out of earlier models continue, this performance will improve.[111]

On the negative side, however, these improvements in emission rates are somewhat below the 90% reduction in emissions legislated in the U.S. Of relevance here is how the in-use emissions of cars will respond over time as older "dirtier" cars disappear. Random tests of in-use vehicles by the EPA[112] show that the shortfall between legislated standards and in-use performance would not only continue, but increase. Crandall et al. (1986) report emission rates in the 1980s of two to four times the legislated levels for HC and CO. Surprisingly, the gap increases as pre-1968 vehicles are retired. In 1982 the levels of HC, CO and NO_x were respectively 30%, 76%, and 5% above those rates that would have prevailed if all cars had met the legislated emission rate. By the year 2000, these levels are projected to be 181%, 486%, and 49% above legislated levels (Crandall et al., 1986). This result follows from the expected rate of deterioration of emission devices,[113] the composition of the fleet and the introduction of more stringent standards.

Ultimately there is a closer relationship between damages and air quality, than damages and emissions. Therefore information on the affect on air quality of a regulatory program is important to its evaluation. Such data for auto pollution are scarce because very little intensive monitoring of emissions from industrial, utility or mobile sources has been carried by region. As a result it is hard to attribute improvements in air quality to any particular source or program. This is a surprising state of affairs given the cost of automobile emissions control programs.[114]

The available evidence on the relevant air pollutants shows relatively little if any change in ambient ozone at monitoring sites. With respect to ambient CO, there have been significant improvements in the concentration levels in urban areas

between 1975 and 1983, which are likely attributable to automobile control policies.[115]

The conclusion drawn from this information is that the current automobile pollution control program in the U.S. is not very effective.[116] It may also be too costly due to a failure to incorporate differences in regional air quality, for example, in certain rural areas.

U.S. environmental laws (and thus Canadian ones by extension) are designed to solve pollution problems in highly sensitive urban areas such as Los Angeles and Denver. Since such standards are surely not cost effective in most rural areas, it may be more reasonable to allow higher emissions than in heavily populated areas. It is difficult to argue that there are even positive net benefits from such programs in rural regions.[117]

This, along with automobile emission rates which deteriorate quickly with the age of a car, differential regulation problems and the lack of credible sanctions in the program, suggest the need for change in the regulatory approach. Particular attention should be paid to the local nature of damages suggested above.

(iii) Alternative Control Schemes

Several variants of two schemes have been suggested as alternatives for or supplements to the current direction regulation approach:

(1) Limit the use of automobiles in urban areas through restrictions or subsidies to alternative modes of transportation; and

(2) Use externality prices to achieve desired emission levels in new and used cars.

Increased parking fees, larger subsidies to urban mass transit and increased investment in mass transit have all been suggested as ways to induce a shift from the automobile to public transit, with the resulting reduction in pollution, noise and congestion levels in urban areas. Analyses of such schemes (Dewees, 1974; Dewees, 1976; Tietenberg, 1988; Button, 1990) have, however, found them to be largely infeasible for a number of reasons:

(1) The low numbers of commuters who use public transit: Dewees (1976) states that 10% of all urban passenger trips in North America are by public transit. In Toronto this becomes 20%. Button (1990) states that a 5% reduction in motoring in the U.K. would result in a 50% increase in transit ridership and would thus necessitate large investments.

(2) A low cross elasticity of demand between modes: Button (1990) found cross elasticities for peak road traffic in London of .025 with respect to bus fares and .056 with respect to rail fares. As well, he notes that an experiment with free public transit in Rome in the early 1970s had a negligible impact on auto traffic. Dewees (1976) estimates that free transit in North America would increase ridership by 30% which, given urban transit ridership of say 10%, would lead to slightly more than 3% diversion of automobile traffic.

A study of various local pollution policies was undertaken in Boston in the early 1970s. The model incorporated data on traffic flows, aggregate emissions and air quality at 123 receptors (Tietenberg, 1988). The alternative policies considered were a 10% reduction in transit fares, a vigorous program of transit extension and the adoption of 1980 emission standards.[118]

The results suggest a much greater reduction in emissions from the new source standards:

Emissions (gms/sec)	Benchmark	Fare Reduction	Transit Extension	1980 Emission Standards
CO	19609	19343	19497	4022
HC	2755	2711	2744	485
NO_x	934	919	933	401
Annual Costs ($1000)	0	11517	95083	120,000

These results indicate that fare reductions dominate the transit extension but neither improves the environment significantly. It appears that local policies of this type are of limited use and should only be used to respond to special local needs.

A number of studies suggest that externality pricing could play an important role in automobile pollution control policy (Dewees, 1974; Tietenberg, 1990; Baumol and Oates, 1979; Anderson, 1977; Tietenberg, 1988; Mills and White, 1978).[119] Indeed, there are a number of reasons why externality pricing might be well suited to help control auto pollution:

(1) The history of the direct regulation approach to auto pollution has been one of delays and postponements. A charge scheme could induce greater technological change.

(2) As Dewees (1974) and Tietenberg (1988) suggest, the marginal cost of control at present abatement levels is quite steep. Indeed it is relatively constant up to abatement levels of 50 percent and then rises steeply. Charges would be preferable to standards, since with cost uncertainty, the former places an upper bound on costs.

(3) Tietenberg (1990) argues that when comparing charges with a property right scheme . . . "Emission charges work particularly well when transactions costs are high . . . For this reason charges seem a more appropriate instrument when sources are individually small, but numerous (such as residences or automobiles). Charges also work well as devices for increasing the rate of adoption of new technologies and for raising revenue to subsidize environmentally benign projects." (p. 30)

(4) The current regulatory program is not particularly cost effective. Tietenberg (1988) notes that the cost per ton of HC abatement resulting from the inspection and maintenance program in the U.S. was two to three times higher than an equivalent reduction at stationary sources. If the pollutant can be considered relatively conservative,[120] the cost of this appendage to the direct regulation program seems questionable. This suggests that an externality price for mobile sources, if properly co-ordinated with abatement at stationary sources, could yield significant cost savings over the current program. As well, as concern with global pollutants such as CFCs and CO_2 grows, the cost advantages of a single externality price for mobile and stationary sources becomes more evident.[121]

(5) Monitoring requirements for an effective externality pricing scheme are no more severe than those of the direct regulation regime. For example, New Jersey currently tests the exhaust emissions of every car as part of an annual inspection. The test takes about 30 seconds (Baumol and Oates, 1979).

Dewees (1974) is among the initial proponents of an externality tax on automobiles.[122] He, however, rejects the use of a tax based on gasoline consumption, arguing that this might only affect the motorist's choice of vehicles and may not reduce overall emissions. Instead he suggests a charge which would vary with the emissions of a particular pollutant per mile. This would give the driver an incentive to seek out less pollution-intensive vehicles. The charge could be adjusted according to region or location.

Since an annual measurement of emissions from every automobile may prove to be so costly, the challenge is to find an alternative externality price which avoids these administrative costs, but retains the character of the charge described above. The solution Dewees[123] suggests is first to estimate the lifetime emissions of a model in a particular year and then to levy a charge per unit of the relevant pollutants (lead, HC, CO, NO_x). The pollution rating of a vehicle and thus, the pollution charge incorporated within its price, can change over time.

As well, random tests of cars on the road could be undertaken to observe the deterioration of the pollution control equipment. Any significant change in this for a particular model or engine class could result in an adjustment of the tax.

This scheme provides adequate incentive for technological change by providing a competitive advantage to a manufacturer: the cleaner the car, the lower its sale price. The in-use testing would also encourage the manufacturer to provide a durable product.

Not all vehicles under this scheme would be controlled to the same extent, across or within jurisdictions. Within a jurisdiction, cars with relatively higher abatement costs would abate less and incorporate higher externality price costs within their sale price. In addition, in low-charge jurisdictions more pollution-intensive vehicles would be sold.[124] The monitoring requirements from this scheme are no more severe than under the current regime.

This scheme does not specify standards to which a car must be built; it only levies an externality price which is capitalized into the price of the vehicle.

High-damage regions may wish to levy an additional charge on vehicles at the time of the annual registration of the auto. The charge, which could be based on the emission/mile rates and the odometer reading, would encourage drivers to purchase clean automobiles and to maintain the control equipment. Alternatively, it would encourage owners of "dirty" cars to drive them less, scrap them earlier and consider retrofit abatement equipment. The drawbacks to this scheme are the need for sealed odometers and the increased monitoring costs.[125]

The advantages of this externality pricing scheme are that manufacturers would no longer have an incentive to delay or postpone the introduction of innovations; the perverse incentive effects of differentiated regulations would disappear; and the owner of the car would have an incentive to maintain the control equipment and/or take advantage of the manufacturer's warranty. As well, an externality price differentiated by region, for example, urban versus rural, would likely be more cost effective.

(iv) Economic Incentive Schemes

There are few economic incentive schemes used to control pollution from mobile sources and certainly none in existence which are as comprehensive as the one described in the previous section. Button (1990) discusses a differential annual automobile tax used in Germany. Also used in the Netherlands, this tax is reduced on automobiles which meet the European Community environmental standards. In both countries, cars are classed in one of three categories, "clean cars" (including "catalyst cars"); "restricted-clean cars"; and "other cars" (OECD, 1989, pp. 70-71). The cleaner the car, the greater the tax advantage.[126]

All of these schemes are merely minor elements in a larger package, dominated by the direct regulation approach.

C. Noise Pollution

i) Introduction

The environmental problem which attracts the most attention in urban areas is the loss of peace and quiet, particularly near expressways and airports. As Gillen and Levesque (1990) point out, this is one of the major impediments to airport expansion.

Noise is a public "bad" in much the same way as air pollution. It interferes with the wellbeing of people through no fault of their own. It is a by-product of a productive process and results in the reduction of a scarce and valuable asset: peace and quiet. Because the ownership of the property right to this asset is uncertain, it is treated as a free good. Airports or airlines which use this scarce asset do not incorporate its value into their cost or profit calculations and thus tend to consume too much of it, that is, create too much noise.[127]

The damage created by noise includes loss of sleep, loss of relaxation, interference with conversations and interference with television reception. While it is also possible to attribute hearing loss to loud noise, this seems to be an unlikely outcome from the type of ambient noise considered here. It is more likely a concern of workers at noisy industrial work sites. Thus the chief impact of noise on those living near airports or expressways involves annoyance rather than health damage.

Noise pollution around airports has grown worse in recent years because deregulation has substantially increased airport activity.[128] This along with continued urban growth has exacerbated the problem.

Despite this, the economics literature dealing with noise pollution (Nelson, 1978; Alexandre, Barde and Pearce, 1980; Harrison, 1983; Button, 1990; Gillen and Levesque, 1990) remains somewhat underdeveloped when compared to that dealing with other environmental problems. While the noise pollution literature exhibits some similarities with the pollution literature reviewed earlier in this report, there is a significant difference in the approach developed by at least one author. Harrison (1983) argues that, in essence, what amounts to a first-best Pigouvian tax can be used in the case of noise pollution at airports. This appears to be one of the few areas in the environmental literature where the first-best scheme is of direct relevance.

This section describes the regulatory approach which currently dominates in North America, reviews some alternative economic incentive schemes suggested in the literature and summarizes noise pollution charge schemes in use at present.

ii) The Current Regulatory Approach to Noise Pollution at Airports

Regulations to reduce aircraft engine noise have been introduced by both the U.S. Federal Aviation Administration (FAA) and the International Civil Aviation Organization (ICAO); these regulations have had a significant impact on the operation of aircraft in Canada.

On December 1, 1969, the FAA introduced the Federal Air Regulations (FAR) 36, requiring new commercial planes conform to certain prescribed standards with regard to noise emissions.[129] These standards are set for take-off, landing and sideline noise. Aircraft not meeting the standards will be denied certification. As well, in 1976, jet aircraft in service were required to meet these Part 36 standards either by retrofit or replacement.

In 1970, ICAO adopted Standards and Recommended Practices for Aircraft Noise (Annex 16). Member states, such as Canada, who accept the provisions of the Annex incorporate its requirements into their own national legislation.

Similar in tone to Part 36, Annex 16 was first applied on January 6, 1972 to provide noise certification standards for jets. Different provisions apply to different planes depending on weight and time of production. New planes are subject to stricter standards. Annex 16 is the basis for certification of airplanes in Canada.[130]

The noise standards imposed on aircraft in North America represent a classic application of the direct regulation approach. Here, as with other environmental problems discussed in sections IV(B) and V(B), this approach has been characterized by delay, postponements and modifications, especially in the U.S. The major concern of airlines has been the extension of Part 36 standards to the existing fleet.[131] Also referred to as the retrofit rule, it requires retrofit of Stage 1

aircraft within a specified period. Opposition to retrofit is noted by Harrison (1983): "Through the Air Transport Association, the airlines have steadfastly opposed the Part 36 extension, arguing that the benefits are not commensurate with costs, that the compliance schedule was impossible to meet (because of the limited availability of retrofit kits and the airlines' limited resources), and that the requirement would compromise fuel efficiency and safety objectives . . ." (p. 64)

Based on such arguments, Congress extended the schedule for planes with two or three JT8D engines, an interesting development since these are the most cost-effective engines for retrofit (Harrison, 1983). This compromise was chosen in order to reduce the impact on smaller airlines, but obviously did so at the cost of efficiency.

iii) Alternative Control Schemes

Several alternative charge schemes have been suggested in the literature. Nelson (1978), derives a set of landing charges[132] designed to induce retrofit. He first determines the cost of retrofit per landing or per unit of noise emissions (EPNdB) reduced and uses this to define the relevant tax or externality price. For example, assume an airline is charged a noise fee for every landing of every non-retrofit aircraft. P, the charge per landing, that induces retrofit by the relevant aircraft types, for example, B-707, DC-8, B-727, B-737, DC-9 can be defined using:

$$P.L = C \,/\, \sum_{t=1}^{n} d_t$$

where L is the number of landings per year; C is the present value of retrofit costs; $d_t = 1/(1 + r)^t$; r is the rate of interest; and n is the remaining economic life of the plane. Therefore,

$$P = \frac{C}{L}\left\{\frac{r(1+r)^n}{(1+r)^n - 1}\right\}$$

Thus, using sound-absorption material (SAM) on a B-707 which costs \$1.417 million in 1975 dollars (Harrison, 1983), assuming a 10-year economic life, with 1000 landings per year and a 10 percent discount rate, P becomes \$231.[133]

Nelson finds that these charges vary from \$301 for a B-737 to \$35 per landing for a B-727 or, in terms of the implicit cost per EPNdB reduced, from \$4.43 for a DC-10 to \$22.67 for a DC-8. Thus a charge of \$25 per dB of noise emissions in excess of the legal standard (106.3 EPNdB for a B-707) would induce retrofit,

because a non-retrofit B-707 has an average noise level of 116.8 EPNdB. This would result in a landing charge of:

$$\$25(116.8 - 106.3) = \$262.50$$

The discounted cost per landing of retrofit for a B-707 was shown above to be $231.

The scheme presented by Nelson (1978) is similar to the noncompliance penalty discussed in section IV(G).[134] Its purpose is to introduce some flexibility into the standards scheme. Airlines, for example, may chose not to retrofit older planes which will be retired in the near future. This may in the long run prove to be cost effective. If older planes are retrofit now their economic lives may be increased. Alternatively, under this noncompliance penalty, newer, quieter planes may be adopted sooner. While in the short run, noise may not be reduced as much under this scheme, the long-run impact may involve less noise at a lower overall cost.

This scheme does not result in airlines delaying retrofit when it is a feasible option. Also since the charge is based only on retrofit, it really requires little or no noise monitoring. Planes which have installed retrofitting do not pay the charge, those that have not, pay a charge which is related to the economic benefit to them of noncompliance.

However, this scheme does have several drawbacks. There are a number of ways of reducing sound damage: reduction of noisiness of aircraft engines, changes in aircraft landing and take-off procedures, changes in the timing of aircraft operation, soundproofing homes surrounding the airport, installing sound barriers between aircraft on the ground and homes, moving people away from airports, etc. Since noncompliance penalty deals with only one of these control techniques, it is unlikely to be cost effective, to provide any more incentive for technical change than the standard approach (see section IV(E)).[135]

Harrison (1983) discusses a different approach to the externality pricing of noise pollution. He argues that there is sufficient evidence to suggest that the airport noise damage function is linear over the relevant range.[136] This condition, if correct, would allow the use of a theoretically correct Pigouvian tax.[137]

Harrison offers other reasons for at least an experimental use of charges to control noise pollution. Unlike other environmental problems, the benefits of noise reduction do not seem to require a determination of the value of life and limb, since health effects are not a relevant component of damages from ambient noise. As well, in the area of noise at airports monitoring does not appear to be a major problem. Indeed, with the exception of Dorval Airport in Montreal, all major Canadian airports already have monitoring facilities (Gillen and Levesque, 1990).

To impose this externality tax, the marginal damage of noise or the marginal benefit of noise reduction must be determined. As Harrison points out, the information required to do this is already available in a number of hedonic house-price differential studies.[138] When people purchase a house, they also purchase the amenities and disamenities of the neighbourhood in which it is located. Given two identical homes, A and B, alike in all respects except that A suffers from the noise generated at an adjacent airport, it is expected that in equilibrium the price differential between A and B will represent the capitalized annual value of damages caused by this noise.[139]

In a summary of 13 empirical studies of aircraft noise and housing prices, Nelson (1978) finds that the mean noise discount of housing prices is 0.62% per unit increase in NEF and ranges between 0.4 to 1.1% in these studies. The annual present value of these damages can be derived by applying a discount factor. For example, a $100,000 home when situated near an airport would fall in value by $100,000 (.62%) = $620. The annual damage caused by the annoyance of noise is $62 assuming a discount rate of 10%.

A study by the Council on Wage and Price Stability (COWPS), reported in Harrison (1983), estimates that in 1975, noise-associated damages near airports in the U.S. amounted to $325 million/year. These results are based on the Nelson study, as well as estimates of the number of people in the U.S. exposed to airport noise levels above NEF30.[140] Assuming that 98 EPNdB corresponds to NEF30, using data on the number of airplane operations by various types of planes and calculating the resulting noise levels, it is possible to determine the number of EPNdB's emitted above the threshold. In 1975, this was approximately 45 million. As a result, a uniform national charge of $325/45 = $7/EPNdB would represent marginal damages from noise.

Harrison points out a number of problems with a uniform national charge of this type, not least of which is that it does not provide a good proxy for the marginal damages of noise. The uniform scheme assumes all EPNdBs cause the same annoyance. There is evidence, however, that the damages of an additional unit of emissions will vary with airport location, time of day of the emissions and even the level of EPNdB emissions.

It is noise exposure with which we are ultimately concerned, not noise emissions. The best basis for setting a noise charge, as well as measuring noise exposure, is NEF levels. If an aircraft taking off from airport X increases NEF levels on average by 0.5 for 1000 households, the airline would be charged for 500 NEF-households. The problem with this measurement is the complexity of the calculation.

An alternative suggested in Harrison (1983) is to devise a formula that incorporates these major variations. He presents various average noise charges for selected U.S. airports based on a general form:

$$C = F(EPNdB, t, A)$$

where C is the noise of charge; t the time of day and A the airport. "The schedule . . . is based on a doubling of the area exposed to a given noise level with every increase of 5EPNdB,[141] a twelvefold penalty for "nighttime" operations (10:00 p.m. - 7:00 a.m.), and separate calculations of the noise damage at each airport . . . " (p. 75).

Two major limitations of the retrofit approach are that all aircraft must comply, despite the fact that for some planes compliance may not be cost effective and that the retrofit scheme concentrates on only one way of reducing noise exposure to the exclusion of others, as did the noncompliance penalty discussed earlier. Both the uniform charge and the noise charge formula overcome these weaknesses. Since, based on Nelson (1978), the cost of retrofit between different types of aircraft varies by a factor of six, it is clear that these charges can increase cost effectiveness. Airlines have the choice of retrofitting or not, or choosing some other means of abatement, such as operational changes.[142]

It is also expected that the noise charge formula will be more cost effective than the uniform charge because there are certain methods of noise control which the firm will only be induced to consider under the formula. These include the reduction of night flights and alteration of airports.

The direct regulation approach is often criticized because it provides little or no incentive to adopt innovations which reduce emissions below the standard. It may even provide perverse incentives to produce costly, ineffective technology due to the pressure of deadlines. Harrison (1983) suggests that this has affected innovations in the auto industry: "Automobile emissions are often cited as a case in point. Many commentators believe that the 1975-76 deadline established in the 1970 Clean Air Act for a 90% reduction in emissions forced the automobile companies to choose high-cost, unreliable, "bolt-on" technology rather than a lower-cost technology that would not necessarily have been ready for production for the deadline." (pp. 105-6)

An effluent charge might have resulted in a durable technology which might have achieved abatement in excess of the 90 percent goal.[143] Similar problems can be envisaged resulting from the retrofitting regulatory approach to airplane noise control and its forced deadlines.

As discussed in section IV(G), the revenues derived from these charges provide one of the main obstacles to their acceptance by polluters. Despite the cost

savings which would accrue under the charge, the tax payments made by airlines would probably wipe out these savings.

It has been suggested that these charge schemes are less likely to be opposed if the revenues are earmarked for noise-related purposes. As Harrison points out however, such uses may lead to expenditures, such as soundproofing homes or compensation for those close to airports, which are ineffective, wasteful and inequitable. As well, often these programs are designed merely to dispose of the revenues generated. Several studies including Harrison (1983), show that soundproofing residences is not as cost effective as alternative abatement techniques and it is difficult to determine on the basis of reasonable equity criteria who should be compensated for noise pollution from an airport. Recent arrivals to the area likely purchased their home at a discount because of the ambient noise in the neighbourhood: the real loss was probably suffered by the original owner of the home.

Although it has not been discussed in the literature, the charge approach presented by Dewees (1974) or the smog tax approach discussed in section VB(iii) could also be applied to aircraft. While not as desirable as that discussed in

Harrison (1983), if monitoring is not easy or inexpensive, these schemes could provide a reasonable second-best solution.

iv) Economic Incentive Schemes

A number of countries have adopted noise effluent charges at airports, including France, Germany, Japan, the Netherlands, and Switzerland (OECD, 1989). Virtually all of these charges are an attempt to compensate those who live in the vicinity of airports.[144] The Japanese and Dutch charges, used ultimately for soundproofing homes and relocating families, are set at a rate to recoup the cost of these noise impact-reduction measures. The Dutch charge varies by type of plane and time of day but is not set high enough to induce airlines to install noise reduction equipment (Kneese et al., 1977).

The OECD has suggested the implementation of a hypothetical noise charge. The OECD scheme, unlike any of the existing ones, incorporates incentive effects. A standard for each type of jet aircraft is set below the level which could be achieved with the best available control equipment. Each jet is also given an uncontrolled-noise rating. Each time the plane lands, it must pay a charge based on the amount by which its noise emissions, measured in dB, exceed the standard. The noise emissions are determined by the type of aircraft and the noise reduction equipment installed. Designed to induce further innovation, the charge is set high enough to induce the installation of the best retrofit equipment available.

The revenues from the OECD scheme are to be used to compensate the victims of noise pollution directly. Presumably they are allowed to decide how to spend this money (Kneese et al., 1977).

It is interesting to note that nearly all the real-world schemes, as well as the hypothetical OECD scheme, are designed in some way to compensate the victims of noise pollution. This runs counter to the recommendations of Baumol and Oates (1988) who argue that ". . . if all the neighbours of factories were paid amounts sufficient to compensate them fully for all damages . . . obviously no one would have any motivation to locate away from the factory" (p. 23). Their opposition to compensation does not, however, rule out a re-allocation of the revenues received from charges. It only suggests that such a re-allocation should not be based on the damage sustained by victims of pollution.

Victims of noise pollution can take action to reduce exposure. With a Pigouvian tax, the optimal charge is levied against the polluters--in this case airlines--and no subsidy is paid to the victims. The result under certain conditions is optimal. This says little about equity. The victims will, in this solution, undertake the optimal amount of defensive action. If they are compensated for annoyance, their incentive to do something for themselves is weakened or even destroyed.

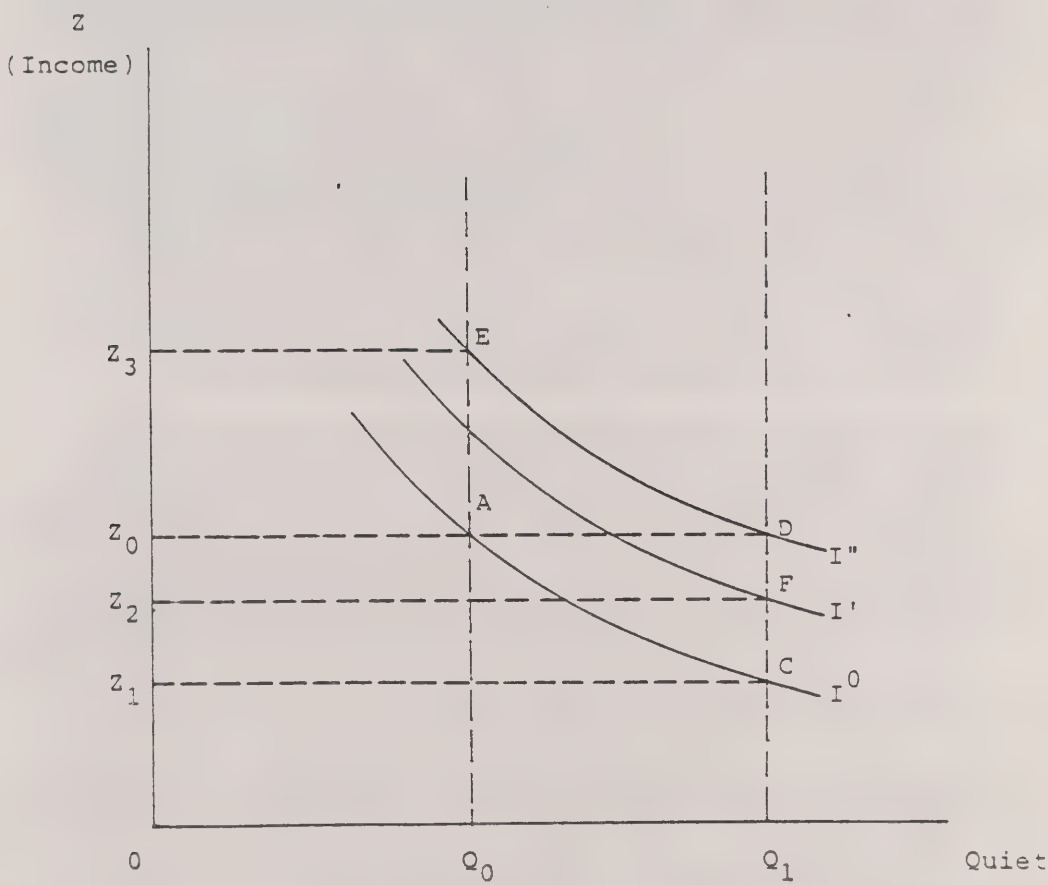
The problem is similar to the moral hazard problem discussed in the insurance literature. If victims are fully compensated for a theft, they have little or no incentive to allocate resources towards burglar-proofing their homes.[145]

A simple example can help to explain the distortion which compensation may induce. Assume an individual lives close to an airport and is annoyed by the resulting noise. Assume there is some action which the victim can take which will totally alleviate the annoyance, that is, result in quiet, Q_1. He or she will of course only undertake that action if the reduction in damages exceeds the costs of the action. Assume the individual does nothing: then the individual, whose tastes and preferences are represented in Figure IX, will be at point A, consuming Q_o quiet and with income Z_o. The noise exposure-control technique available must cost less than $Z_o - Z_1$ or the victim will not adopt it. Assume it costs $Z_o - Z_2$ per period. The victim will adopt this device since it will raise his or her utility from I^o to I'.

Now assume the government decides to compensate the individual for damages suffered. These damages, with a monetary value of $AE = Z_3 - Z_o$, would make the victim as well off as he or she would have been had there been no noise.

Provided the victim expects this compensation to continue, he or she has no incentive to undertake any defensive action to alleviate the noise.[146] This is likely not an optimal result.

FIGURE IX
THE IMPACT OF COMPENSATION FOR A VICTIM OF NOISE POLLUTION



Burrows (1980) takes issue with this result on grounds of justice. He argues that to allow "optimal" levels of pollution with no compensation to the victims may be unjust. On the other hand, to reduce pollution levels to zero, or to compensate victims may be inefficient. Thus a just and equitable solution may involve some compensation and a resulting loss in efficiency. Burrows (1980) suggests that it may be possible to limit these losses:

> In practice it will be difficult to distinguish between cases of pollutees who, despite taking care, are suffering high damage levels owing to their own sensitivity, and those who expose themselves to such damages knowing they will qualify for more compensation. The result may be a need to approximate full compensation by defining "reasonable" sensitivity and "reasonable" care taken to avoid exposure, and basing uniform payments to pollutees on the damage associated with reasonable pollutee behaviour. This is, of course, exactly what the courts have in mind when they relate damage awards in tort cases to the loss that would have been suffered by the "reasonable man." (p. 105)

Endnotes

1. In the environmental economics literature, externality prices are generally referred to as "emission fees" or "effluent charges" (EC). The latter term will be used extensively in the remainder of the paper.

2. There are various categories of externalities discussed in the economics literature. The first distinction is between a pecuniary and technological externality: a pecuniary externality arises when an agent's actions cause a price to change affecting the welfare of others. But since there is no direct effect on real variables this does not really create a misallocation (Baumol and Oates, 1988). All of the examples given above, and indeed all relevant examples within the context of this study, are technological externalities. Technological externalities can also be divided between external diseconomies and external economies. In the former case the agent imposes a cost on others, whereas in the latter, a benefit is imposed. All of the examples discussed in this study are external diseconomies. Finally, Baumol and Oates (1988) draw a distinction between depletable and undepletable externalities. We are concerned only with undepletable externalities, in which the consumption of a "bad", i.e. something which reduces an individual's utility as, for example, air pollution, by one individual does not reduce its availability to others, i.e. it exhibits jointness in supply.

3. Of course if the market failure involved an external economy, the solution would involve a per unit subsidy to the externality generating activity. Our primary concern throughout is, however, with external diseconomies.

4. The ownership, or property right, to this environmental asset is sometimes the source of much debate. If the polluter is forced to pay for his or her use of this asset, then implicitly the property right is vested in the hands of those who suffer from pollution damages. As Coase (1960) points out, however, at least in the small numbers case, where negotiation is feasible, ownership of the asset can be vested in the hands of polluters or those suffering from the pollution. The outcome of negotiations will, in either case, be optimal. Of course the distribution of income will be affected by the ownership of the resource.

   In the large numbers case (which is of most relevance to the discussion here) where negotiation is no longer feasible, ownership of the resource is usually vested with the government. The government must then take action to induce a change in behaviour by the polluter. This action can involve effluent charges, regulations, subsidies or transferable discharge permits. The design of such schemes has definite distributional consequences some of which are discussed later in section IVG.

5. Baumol and Oates (1988) provide a general equilibrium approach to the problem. While their results are richer, the intuition behind them is not always obvious. The simpler approach above provides essentially the same answers but more intuition.

6. Firms maximize profit, and as long as the environment has a zero price the firm will choose not to limit its emissions (or, in other words, its consumption of the environmental asset). It will voluntarily abate only if the emissions have some market value.

7. A more general damage function would be $f(e_1, e_2)$. The form presented above implicitly assumes that the location of the emission source is irrelevant to the damages created, i.e.

$$\frac{\partial TD}{\partial e_1} = \frac{\partial TD}{\partial e_2} .$$

This is consistent with the assumptions of a perfectly mixed environment and a conservative pollutant, i.e. a pollutant which does not degrade over time. Under such conditions, the source of an emission point in an airshed would be irrelevant. The only important variables in determining damages are thus the rates of discharge from the point sources. Certain global pollutants, such as ozone depleting CFCs, can be characterized in this fashion. If the pollutant were not conservative, and hence were degradable, the damages at the receptor would depend not only on the emission rates, e_i, but also on the location of the emission sources.

In addition, it is initially assumed that $d^2f/de^2 = 0$, which implies that the damage function is linear. This last assumption is tantamount to assuming a constant marginal damage of emission function. Thus $df/de = c$. These assumptions will be altered as the analysis proceeds.

8. Assuming that a firm maximizes:

$$\pi_1 = R(Q) - C(Q) - C^1(A_1) - ce_1$$

where R is revenues, C is production costs, Q_1 is output and π_1 is profits. The first order conditions imply:

$$\frac{\partial C^1}{\partial A_1} = c$$

The firm will abate pollution until the charge (c) set by the regulatory agency is equated to the marginal cost of abatement. Note that it is assumed costs are separable in abatement and output. This, however, is not crucial to the results.

9. This result obviously generalizes to the case of n polluters.

10. It is assumed that monitoring costs would be the same under either scheme. Monitoring differences, however, become more important in the second-best world which is discussed later.

11. The direct regulation approach usually involves standards and penalties. Often ambient standards for a particular pollutant are set at various receptor points. To achieve these ambient standards, effluent standards are set for each source. Firms that do not abide by these standards may be subject to fines. These effluent standards may be stated in terms of the overall weight of emissions per unit time, the concentration of emissions, or the weight of emissions per unit of output or input. In the U.S. technological standards are frequently used which define a certain abatement technology required in various uses.

12. A complete description of the derivation of a pollution damage function is provided in Dewees, Everson and Sims (1977, chapter 3). Burrows (1980) also discusses the possibility that the marginal damage function may, in certain cases, be negatively sloped. Such a situation could provide several interesting results to be discussed later.

13. Some have argued that there is still an informational advantage. More precise knowledge of abatement costs are not necessary. An optimal solution could be attained if the authority is able to calculate "current" marginal damages. i.e. marginal damages at current emission levels. The authority would simply adjust the charge to ensure it remains equal to current marginal damages. This will ultimately converge to the optimal solution (Baumol, 1972). As will become obvious, with imperfect market structures, nonconvexities and the difficulty of calculating damages, this advantage is more apparent than real.

14. Thus the environment of the airshed is not perfectly mixed, and the pollutant is a local public "bad."

15. Watson and Ridker (1984) extend the analysis by introducing nonlinearity.

16. In this same vein, Robert and Spence (1976) suggest a scheme involving charges, standards and property rights which they argue, results in economic incentives which more closely approximate the marginal damage function than any one of the individual schemes. The scheme involves the environmental authority issuing a number of marketable pollution permits. The equilibrium price of these, p, is determined in the marketplace. The polluter is allowed to emit pollution in excess of the number of permits purchased but must pay a fee, f, for every excess unit emitted. As well, the authority offers a subsidy, s, per unit of unused permits. They show that in equilibrium,

$$s \leq p \leq f$$

This system leads to an economic-incentive step function which constitutes an approximation to the marginal damage of pollution function.

17. In a study of airport noise control Harrison (1983) argues that the marginal damage of noise in the relevant range is virtually constant. With auto pollution, rapidly rising costs at legislated levels of control has resulted in large welfare losses through overly severe controls in all regions of the country (Tietenberg, 1988). These results suggest the use of externality pricing in these two cases may not be unreasonable.

18. This study uses cost and damage estimates for electric utilities, industrial point sources, residential and commercial sources and transportation sources. In addition to these air pollution sources it also includes various other water pollution sources.

19. This result should not be too surprising to those versed in the theory of second best. This theory basically states that in the presence of multiple market failures (in our case two market failures: pollution and monopoly power) correcting one of the failures will not necessarily result in a Pareto improvement.

20. Note the important distinction that this charge is levied per unit of emissions, not per unit of output. A tax on units of output induces a firm to abate by reducing output. It has no incentive to install abatement equipment or to change operating procedures. Even if these are less costly ways to control pollution, a charge on output would never encourage the polluter to undertake these activities. On the other hand, a charge on emissions will induce the polluter to seek out a least-cost method of pollution reduction whether that involves reducing output, altering operating procedures or installing abatement equipment.

21. Note a first-best optimal solution could be achieved by imposing an externality price on emissions equal to marginal damages and paying the firm a subsidy of GJ per unit of output. This would lead to an output of OQ_o and a price per unit of output of OH. Baumol and Oates (1988) investigate a second-best tax which balances the externality and monopoly costs. If it were not politically feasible to subsidize a monopolist such a tax could be used.

22. An externality price set equal to marginal damages at the optimal level of emissions is often referred to as a "Pigouvian tax" after A.C. Pigou who is often given credit for first suggesting this policy.

23. A nonlinear damage function is assumed.

24. Baumol and Oates (1988) describe a situation in which a firm can benefit by increasing its emissions. By driving away those affected by pollution, the marginal damage of emissions and, hence the tax, is reduced.

25. This means that for given equilibrium prices and hence, a family of budget constraints, a competitive economy will always move to the highest point of tangency between the budget constraint and the production-possibilities frontier.

26. The concept of a potential Pareto improvement is based on the ability of gainers from a policy change to more than compensate losers. Such a change would obviously increase the value of output and is thus based on this proposition linking a Pareto optimal outcome and the maximum value of output.

27. If an extremely noxious pollutant is emitted into a lake it may very quickly kill the fish population. Additional emissions beyond this point may have relatively few additional effects.

28. If $e = e_o$, society would incur losses equal to area A. As e increases beyond e_o, however, $MD_e < MC_A$; thus, there are net benefits from increased emissions until e_2. B represents the net benefits of increasing emissions from e_o to e_2.

29. Baumol and Oates (1988) also state that this is true in the presence of several externalities and provide an interesting example to demonstrate: the automobile is a much heavier polluter per passenger-mile than a railway. Optimal policy might involve subsidizing rail travel and/or taxing the use of the automobile. The ultimate impact might be a large reduction in auto pollution and a smaller increase in pollution from railways. Thus an optimal policy could result in a particular polluter increasing activity and pollution.

30. As Baumol and Oates (1988) point out, an optimal externality price requires knowledge not of current marginal damage (assuming nonlinear damages) but rather of marginal damage at the optimal emission level (see Figure I (c)). "If there is little hope of estimating the damage that is currently generated, how much less likely is it that we can evaluate the damage that would occur in an optimal world that we have never experienced or even described in quantitative terms." (p. 161)

31. This separability has been challenged at least implicitly in a recent paper by Oates, Portney and McGartland (1989).

32. Ambient standards define the allowable concentration of a pollutant in a water body or airshed. Effluent standards define the allowable discharge of a pollutant from particular pollution sources.

33. At least in Ontario, the practical applications of these standards has been based less on the adversary approach and has instead emphasized co-operation and persuasion. For a review of the relevant environmental legislation for Canada see Nemetz (1986).

34. The polar opposite approach to this system of moral suasion might be termed the confrontation approach which would involve strict adherence to available standards. It is not clear that this would be a superior tactic given monitoring and legal costs and the limited budgets of environmental agencies.

35. More current evidence on fine levels primarily related to water pollution is provided by Nemetz (1986).

36. The passage of the Act was supported by manufacturers who feared that following California's earlier lead, each state might pass laws requiring different standards on automobiles.

37. (Mills and White, 1978) (p. 352): "In March 1964, the auto companies told the state that the 1967 model year was the earliest that they would be able to install exhaust control devices. In June 1964, the state certified four devices, all produced by independent part manufacturers. As a result exhaust control devices became mandatory for the 1966 model year. In August 1964, the auto companies announced that they would, after all, be able to provide exhaust control devices, of their own manufacture, for the 1966 model year."

38. Shortly before this extension in 1973, the Honda CVCC engine was certified as having met these standards. (Tietenberg, 1988).

39. On the other hand, it has been suggested that many European models were excluded from the American market due to the more stringent U.S. standards. As well, while domestic manufacturers did withdraw otherwise viable engines from the U.S. market, they continued to sell these models in the Canadian market. Nevertheless, the extension alluded to in the previous footnote suggests that a broad application of this sanction to the U.S. manufacturers was unlikely.

40. Some emission control technology is of course provided by outside suppliers. They clearly are not as inhibited in their innovation as the polluting firms. Their efforts may, however, still be dampened under the direct regulation approach, as is discussed in section IV(E).

41. Basing standards on a unit's date of entry.

42. Since 1979, a joint federal-provincial grant has also been available to pulp and paper mills for modernization and pollution abatement. Some parties have expressed skepticism regarding the possible effectiveness of this scheme (Victor and Burrell,

1981).

43. A simple example can help to explain this idea. If the uniform charge were $5 and the extra or "marginal" cost to the firm, at current levels of abatement, of abating one more unit of pollution is $2, a cost-minimizing firm would abate because this is cheaper than paying the charge. Additional abatement would cost $2 per unit but would save the firm $5 in effluent charge payments, a net gain to the firm of $3. Obviously abatement will be expanded as long as the charge exceeds the incremental or "marginal" cost of abatement.

44. In this case because the overall standard is set per annum the charge would be adjusted each year if it were determined that aggregate emissions of SO_2 were not exactly one million tons.

45. The increased charge could be offset, at least in theory, by technological progress in abatement.

46. It is also possible that the desired standard could vary from year to year. This would also require varying the effluent charge.

47. Biochemical oxygen demand (BOD) is a measure of the reduction in a stream's dissolved oxygen required to break down organic wastes.

48. A cost-effective scheme achieves the overall standard at least cost.

49. Usually the argument is couched in terms of a uniform standard. In reality standards often vary over groups of firms for equity reasons, but are generally uniform within each group of firms. It is doubtful that this complication affects the argument presented above in any significant way. If the variation in standards across groups was based purely on abatement costs then this scheme too would be more cost effective than a uniform standard.

50. This is because the extra or marginal, cost of abatement at all point sources will be equated. If this were not true, abatement could be shifted from high to low-cost sources with a resulting decline in costs. There is empirical evidence provided by Pittman (1981) that under the current direct regulation-enforcement framework in the U.S., pulp and paper mills exhibit systematic differences in the marginal cost of abatement at current loading levels. This may suggest the existence of serious inefficiencies resulting from the current system of pollution control regulation.

51. The models presented in the remainder of this section are based loosely on the presentation in Bohm and Russell (1985).

52. The other terms in equation (7) were defined previously in section II.

53. Note the similarity between this equation and equation (5). If λ were set equal to the marginal damage of pollution, this scheme could yield the first- best optimum.

54. If the marginal cost of abatement at source 1 (MC_A^1) exceeds that at source 2, total abatement costs could be reduced by allocating more abatement to source 2 and less to source 1. For example, assume MC_A^1 = \$7 and MC_A^2 = \$3 at given emission levels, e_1 and e_2. As well, assume $e_1 + e_2 = S$, so that the standard has been achieved. Could this be a least-cost solution? The answer is no. If emissions are reduced at point 2 by one unit, abatement costs at source 2 would rise by about \$3. If emissions at point 1 are simultaneously allowed to increase by one unit, abatement costs at source 1 would fall by about \$7. Overall costs would decline by about \$4 and the overall emission standard would still be met. As long as $MC_A^1 \neq MC_A^2$ costs can be reduced by such a reallocation of emissions.

55. It is assumed that enough polluters are involved to prevent any one polluter from acting as a price-maker.

56. The literature usually assumes that in the absence of such information the authority imposes a uniform reduction on the polluters. For example, each firm might be required to abate:

$$\frac{\{Z_1 + Z_2 - S\}}{2}$$

This scheme would only be cost effective if each source had identical abatement cost functions.

57. If this cost turns out to excessive the scheme may require some knowledge of abatement costs so that the initial charge can be set in the neighbourhood of λ.

58. If the value of α changes seasonally, then the charge should be adjusted seasonally.

59. This problem is written explicitly as:

$$\min C^1 + C^2 - \lambda_1 [S_1 - \alpha_{11}e_1 - \alpha_{21}e_2 - B_1] - \lambda_2 [S_2 - \alpha_{12}e_1 - \alpha_{22}e_2 - B_2]$$

where λ_1, and λ_2 are Lagrange multipliers.

60. This is especially true for global pollutants such as carbon dioxide or CFCs where the location of the pollution source is not of great concern.

61. The Atkinson and Lewis study deals only with particulates. Spofford et al. deals with particulates, SO_2, as well as water pollution. The results presented here refer to a particulate standard of 75 mg/m^3 at the worst receptor and a SO_2 standard of 80 mg/m^3 at the worst receptor.

62. The least-cost solution would be attained by a system of differential charges such as those described in equations (9) and (10). This involves a different charge at every source, and while administratively an unlikely choice, shows the maximum gain from an economic incentive scheme.

63. These effluent charge payments are not a cost from society's point of view. They are a transfer from polluters to the government. Indeed if the environment is considered to be a scarce and valuable resource, this merely represents a payment for using its services, no different than a wage paid to a labourer to rent his or her human capital. While this payment is not a real cost to society, it is considered a cost by the firm and thus has important political dimensions which will be discussed later.

64. Of course no taxes are paid under the SIP, as they are under a SECH. For example, using Figure V(a) and assuming a SECH = λ, the abatement costs imposed on the firm as a result of abating $Z_1 - e_1$ is the area Z_1e_1A. Under a SIP that required this same abatement, the cost would be the same. Under the SECH however, an additional cost of $O\lambda Ae_1$ is imposed on the firm. While this is merely a transfer to the government, and not a real social cost, it is recognized as a cost by the firm. The only true social cost, in terms of resources used, is Z_1e_1A.

65. Spofford et al. also use ZECHs in which uniform charges are set within zones but vary across zones. They consider a 3- and 11-zone system and find that costs of effluent charges are once again less than under the SIP. For the 3-zone system, the abatement costs under ZECH are 91% of SIP costs, whereas with 11 zones this becomes 66%. When effluent-charge payments are included, firm costs are higher for the charge scheme.

66. In reality, such information is not available to the authority and, to reduce administrative costs, programs such as the SIP and SECH investigated here are likely to be used.

67. The payments under a SECH would be $OECAe_1^s + OECDe_2^c$.

68. A similar result using a more sophisticated model is described in Russell (1986).

69. Atkinson and Lewis find that effluent-charge payments are over three times higher than abatement costs for particulates under a SECH. Spofford finds a factor of 3.9 for particulates and 1.0 for SO_2. More recent studies by Seskin and Anderson (1983) and Krupnick (1986) find payments under a SECH to be from one to two times the annual control costs for NO_2.

70. In the Atkinson and Lewis and the Spofford et al. study, the least-cost solution exhibits abatement costs from 3% to 5% of SIP costs. Only the Atkinson and Lewis study presents results inclusive of charges for the LC solution. Since that study, firm costs become 15% of SIP costs, it provides no empirical evidence of the theoretical result provided here.

71. If these charge payments were redistributed to the firms in some lump-sum fashion such opposition might be tempered.

72. The comparison presented in Tietenberg (1990) is between abatement costs in the LC case and the direct regulation case. He finds that costs under direct regulation are from 1.07 to 22 times higher.

73. In figure VI, this is represented by the excessive abatement at source 2 under the SIP and particularly, the SECH scheme.

74. Oates et al. (1989) find that a standard of 90 mg/m^3 under an incentive scheme leads to an actual concentration of 61.6 at their receptor 1. The direct regulation approach leads to a similar concentration at this receptor for a standard of 100. This will of course not occur at critical receptor points.

75. Externality pricing is not widely used in North America but where it is used, for example in Germany and the Netherlands, there is no evidence of major short-term dislocations. These European schemes are, however, characterized by subsidies to waste treatment investments (Brown and Johnson, 1984).

76. This idea is at odds with the use of this revenue suggested by others (Tietenberg, 1990, for example).

77. Whalley and Wigle (1990) consider how different countries would fare under a carbon tax designed to reduce CO_2 emissions by 50 percent between the years 1990 and 2030 in various regions of the world. The North American tax was found to yield a tax revenue over this period with a present value in 1990 of $12 trillion. If the relative share of carbon production in Canada and the U.S. remains the same, this would generate revenues for the Canadian government of around $1 trillion, a substantial gain from externality pricing.

78. Lee and Misiolek (1986) take this desirable non-neutrality of effluent charges into account to define an optimal charge. For example, raising the tax causes a reduction in emissions and, under certain conditions, increases revenues, thus reducing distortions. It might therefore be expected that the optimal tax should be higher than usually suggested. The authors show however, that an increase in the externality price may also decrease tax revenues (depending on the elasticity of emissions with respect to the tax). Thus the optimal effluent charge may be either higher or lower than the Pigouvian tax.

79. Bohm and Russell (1985): "The extra push toward the development of new production and discharge reduction technology provided by these instruments seems likely to dwarf in importance the short run, and to some extent illusory, advantages to be gained by specifying actions or stigmatizing pollution at any non-zero level . . ." (p. 455).

80. The analysis also evaluates auctioned property rights, property rights initially distributed free of charge and a subsidy per unit of pollution reduction. The analysis presented in this section provides a comparison only of the effluent charge and direct controls.

81. It is assumed in the above analysis that the innovations cannot be patented. Milliman and Prince (1989) find that allowing patenting does not alter their conclusions regarding the ranking of policies. While the assumption of non-patented innovations seems unrealistic, there is in fact a precedent in the auto industry. In the mid-1950s the auto manufacturers signed an agreement guaranteeing equal access for all manufacturers to each others' control patents on a royalty-free basis (Mills and White, 1978).

82. This second-best scheme would be adopted for reasons discussed earlier in this report, not least of which might be that information on the marginal damage function is limited.

83. The incentive for innovation under direct regulation is probably even less than the above analyses suggests: first, the above models do not incorporate the differential regulation which characterizes most real-world direct regulation programs. The setting of standards that are stricter for new polluters has extended the life of old technology. Mallory (1990) reports that plants in the U.S. which would have normally closed in 1995 will, due to this form of regulation, stay in operation until 2003. Second, direct regulation at the moment involves design standards that mandate particular technology and, in particular, often require "best available technology." The invention of better control equipment may force adoption, regardless of the cost of equipment already in place.

84. By setting dynamic efficiency as a goal it is assumed, implicitly, that technological change is desirable. Like any activity, however, there is an optimal amount. Allocating infinite amounts of money towards technological change in pollution control makes little

sense. To argue for the adoption of a policy which encourages technological change in this area and to assume that the most desirable policy is the one which yields the greatest incentive to technological change, implies that research and development expenditures on abatement equipment would be sub-optimal in an uncontrolled situation. This seems a reasonable assumption.

85. Harford (1978) shows that the optimal level of pollution is that amount at which pollution damages, control costs and the costs of enforcement are minimized. Thus ". . . the damages caused by one more unit of waste should be greater than the costs of physically eliminating the unit by the amount of the extra enforcement induced by the increase in tax evasion caused by the additional pollution tax required to reduce pollution by the last unit." (p. 41)

86. Linder and McBride (1984) discuss the stochastic nature of emissions and the trade-off alluded to here between convicting the innocent and not prosecuting the guilty.

87. Monitoring is even simpler in those circumstances in which the operation and maintenance of equipment have little influence on emissions. An example is SO_2 emissions from fossil fuel boilers: knowledge of the sulfur content of fuels burned and the sulfur collected will provide a reasonable estimate of emissions. Such a procedure would not work for NO_x however, since such emissions depend on variables such as furnace temperature and efficiency.

88. Harford (1978) shows that polluters reduce emissions in response to increased probability of detection and increased fines.

89. It is ludicrous to argue that this suggests favouring ineffective policies.

90. If the analysis of section IV(B) is correct almost any effective environmental program would be a source of greater dislocations.

91. A comparison of charges and standards can be through a positive or normative analysis. A normative analysis explains which environmental policy *should* be adopted. Much of what has been stated to this point in this study is basically normative in character. An alternative, but related, type of analysis tries to explain why a particular policy dominates others and why it *is* used more extensively. With respect to environmental policy, one of the first positive analyses of charges and standards can be found in Buchanan and Tullock (1975). Quinn (1983) and Dewees (1983) extend this analysis.

92. Quinn (1983) discusses compensation for those who lose from environmental policies. He specifically identifies labour, consumers and polluters, but not the victims of pollution. In fact, Baumol and Oates (1988) show that Pareto optimality in the face of an externality requires only a Pigouvian tax (or externality price) imposed on the polluter. Optimality, in fact, rules out compensation for those who are the victims of pollution, since it may induce supra-optimal numbers to ignore the externality costs and move close to the source of the externality. Burrows (1980) however, takes issue with this and argues that a just solution, although it may not be compatible with an efficient solution, should compensate the receptors of pollution.

93. In essence, under the current system some individuals can use the environment free of charge thus imposing costs on others.

94. Implicit is that the damage of pollution is linear and that pollution per unit of output is constant. Thus the change in damage resulting from a change in output is constant and this tax is the optimal Pigouvian tax. Dewees (1983) also presents a model designed to investigate the political feasibility of alternative policies. His model allows emissions per unit of output to vary which is a far more realistic assumption.

95. Dewees (1983) considers the more realistic policy of differential standards. This grandfathering policy allots stricter standards to new firms, thus raising their costs by more than older firms. Dewees shows that this policy creates an implicit barrier to entry and thus economic profits for old firms. It is clear why existing firms and labour support such differential environmental policies.

96. Coelho (1976) provides an insightful modification of the Buchanan and Tullock result. He argues that negotiations between firms will take place since q° in diagram (c) is not a stable output. He argues that through merger or purchase, the number of firms will decrease and average firm output will move to q' in diagram (c). This is equivalent to the introduction of marketable quotas. This is beneficial to firms and society (assuming no increase in market power which may negate at least some of the social benefits) since unit costs of production will fall. Coelho argues that firms will exert political pressure to allow such quota exchanges. Dewees (1983) comes to a similar conclusion.

It is interesting to note that the U. S. environmental system has introduced a limited market in quotas in non-attainment regions where entry and expansion of firms have been prohibited. For more on this scheme of offsets, bubbles and netting see Tietenberg (1985). Coelho's marketable quotas do not, however, lead to an efficient result since unit production costs in diagram (c) still exceed q_iC.

As well, if the quota scheme were to increase market power for firms already in a region, then these marketable quotas may take on greater value to the firms in the region than to new firms in more competitive industries who would like to expand into that region. This suggests that marketable quotas would be used for expansion of existing firms and would not generally be traded externally. Hahn (1989) reports that of

over 2000 offset trades in the U.S. to date, less than 10 percent involved external trades. (New sources entering a nonattainment region, i.e. a region that has not attained its designated ambient air quality, or existing sources which wish to expand, must offset their potential new emissions by inducing already existing sources to reduce their emissions. This is the basis of the offset scheme.)

97. Industry often argues that charges represent double taxation: inducing abatement costs and requiring a payment for the remaining emissions. As well, some environmentalists consider these charges to be a license to pollute. Large firms would continue to emit waste and just pay the charge which is then passed along to its consumers. Obviously these two objections are at odds. The available evidence, discussed earlier, suggests that appropriately set charges will be effective.

There are also objections to the philosophical idea of an externality price, especially with respect to pollution. Some people look at polluting, at any level, as an immoral and illegal act. Allowing people to carry out an illegal act provided they pay a charge is therefore immoral. Most economists look at polluting as a rational economic act that results because of a failure in the market mechanism, not a failing in the moral fibre of society.

98. Dewees (1983) argues that existing firms may prefer differentiated direct regulation to no controls at all. Tietenberg (1988) reports evidence that implies that in the U.S. tissue industry ". . . In the long-run, existing plants were actually more profitable than they would have been without the regulation, because the regulation served to diminish competition from new sources." (p. 463) He also suggests that U.S. environmental legislation has reduced competition, in some regions, in smelting.

99. All but the staunchest advocates of externality pricing look upon such schemes in conjunction with certain forms of direct regulation. Anderson et al. (1977) while supporting the usefulness of externality pricing, suggest that there are situations in which direct regulation may be more useful than economic incentive schemes. For example, in a situation in which the authority had good knowledge of abatement costs, there are few pollution sources, and the target emission level for some particularly hazardous waste is zero.

Baumol and Oates (1979, chapter 20) suggest that direct controls are advisable when rapidly changing environmental conditions may be heading toward an environmental catastrophe. Such conditions may not be predictable, thus making adjustments in, or reaction to, an effluent charge uncertain. In addition, it is certainly not desirable to set charges permanently at a level high enough to guarantee environmental quality, under all conditions. Direct controls appear to have an advantage in such extreme situations since they can incorporate the option of shutting down certain sources, e.g. incinerators, when air quality falls below some predetermined level. Such a direct control could be used with an effluent charge, the latter being considered a long-run measure and the former to control dangerous short-term fluctuations in the assimilative capacity of the environment.

100. The OECD (1989) report on economic incentives states that the only significant imposition of effluent charges is found in the schemes used to control water pollution in France, Germany and the Netherlands.

101. These are usually design standards which specify the abatement equipment which must be installed.

102. The highest effluent tax rates in Europe are in the Netherlands. There is some evidence these charges have been effective (Tietenberg, 1990), but since they are used in conjunction with direct regulation, it is difficult to determine the relative incentive effects attributable to either scheme. It has been estimated that over the past 15 years, water pollution has declined by 90 percent in the Netherlands (Hahn, 1989).

103. The authority usually asks the firm to submit a compliance schedule stating what abatement steps will be taken and when. The authority will then accept or revise this document and issue a compliance order. A civil penalty may be imposed on firms that do not comply with this request.

104. Emission charges have also been imposed to a limited extent on air pollution in France and Japan and on noise pollution in Japan and various European countries (OECD, 1989). The noise schemes are designed primarily to compensate those who are disturbed by noise near airports. The French air pollution charge revenues are returned in subsidy form to those paying them. The Japanese air pollution charge is used to compensate victims of pollution. These schemes have virtually no incentive effect and have little or nothing in common with externality prices. In short, they have little economic significance.

105. For a similar view with respect to general environmental problems see Chant, McFetridge and Smith (1990). They state that environmental problems could be viewed as a result of "government failure," since it is government which fails to assign appropriate property rights. This failure often occurs because governments attempt to address other social goals, e.g. regional equity, full employment, which may conflict with the environmental problem. For a good example of this conflict related to electricity generation in the U.S., see Chant et al. (1990).

With respect to transportation, Button (1990) suggests that current policy may be a result of regulatory capture: a process first discussed by Stigler (1971) in which the regulatory authority adopts policies which are beneficial to selected politically powerful groups, rather than to society at large. Button argues that much of British externality policy related to transportation, has attempted to minimize the impact on transport users. This thinking is consistent with the positive theories of environmental regulation discussed in section IV(G) of this study.

106. It is of course possible to monitor the noise from traffic in particular areas. Such measures are analogous to measuring ambient air quality in air pollution problems. The use of externality pricing, however, requires, at minimum, that each decision-making unit's contribution to this ambient level of noise be determined. Only then can the appropriate authorities design a policy which imposes the appropriate costs on those creating the problem. Traffic noise is analogous to other non-point source pollution, e.g. agricultural run-off, where it is impossible to identify the exact source of the deterioration in the ambient environment. The use of an externality price would require a detailed log of the path of each trip and its time for each automobile. The charge per trip might be adjusted for time of day, location of path and the level of noise emissions from the vehicle. Of course, a scheme for automobiles or trucks requiring this amount of information would not likely be cost effective. This does not imply that public policy should not be used to address the traffic-related noise pollution problem, but only that, with current technology, an externality price is not practical.

107. Since very little of a substantive nature has been written about the Canadian experience with direct regulations, much of what is presented here on the efficacy of the current regulatory approach is based on U.S. evidence presented in Crandall et al. (1986). The discussion is relatively general, emphasizing broad lessons from the U.S. regulatory situation that may be applicable to Canada. Nemetz (1986) notes after pointing out that the transportation sector in Canada had successfully reduced CO, HC and NO_x emissions during the early 1980s, that ". . . such an achievement would have been very difficult if not impossible were it not for parallel American activities in this regulatory area over the same period." (p. 558) This closeness of the Canadian and American approach suggests that lessons from the American scheme should provide interesting insights vis-à-vis policy in Canada.

Because of the closeness of the Canadian and U.S. auto industries, parallel development is expected, but until recently there have been some differences. In particular, the 1984 report of the House of Commons Subcommittee on Acid Rain noted that critical Canadian standards for HC, CO and NO_x were three to seven times less restrictive than U.S. equivalents.
These differences likely reflected a belief that U.S. standards were too stringent, especially in Canadian circumstances. This position appears to have changed with bilateral acid rain negotiations and, as a result, in May 1985 the Canadian government announced new standards, to take effect in 1987, which were consistent with U.S. standards.

108. Assume, following Gruenspecht (1982), that:
P_i is the market value of a particular car, i, in operable condition;
SV_i is the scrap value of car i;
PR is the price of a unit of repair;
R_i is the number of units of repair required to make car i operable; and, i is the year and make of the car.
If $P_i - SV_i < PR\ R_i$ then a rational owner will decide to scrap this car. Since new and used cars are substitutes an increase in the price of new cars will eventually affect the price of used cars. P_i will be expected to rise. Assuming that PR, R_i and SV_i are unaffected for various models and vintages, the effect would be to reverse the above

inequality for various models and vintages of used cars, for a longer period of time. Thus a policy, such as stricter environmental controls for new cars, which raises the cost of new cars with no direct symmetric effect on the cost of used cars, will likely reduce scrapping rates.

109. In Canada, the probability of being fined for a poorly performing emissions system is minuscule. The U.S. has, however, in recent years adopted an inspection and maintenance program. This is compulsory in non-attainment regions. Currently about one third of the cars and light trucks are subject to inspection. The system however, is suspect. For more on this see Crandall et al. (1986, chapter 5).

110. There was also concern about damage to vegetation and reduced visibility. Since the initial government response to these concerns in the late 1960s there is a growing belief that the benefits of auto pollution reduction, on which the program was initially based, have been greatly exaggerated. Indeed it has been argued that in the U.S., the costs of the regulatory program outweigh the benefits (see Crandall et al., 1986). Crandall concludes that ". . . our society is very cautious about potential undiscovered health effects or that the emissions control program for automobiles is too stringent." (p. 115) As well, Tietenberg (1988) states that ". . . The conclusion that the costs of control exceed the benefits of automobile pollution control seems to be generally shared . . . It is nonetheless interesting that because the current policy forces manufacturers to operate on a very steep portion of the marginal control cost function, this benefit uncertainty does not seem to affect the conclusion that the current standards are inefficiently strict." (p. 394)

111. This improvement is clearly the result of the regulatory program, rather than some exogenous technological change. The evidence is provided by the Canadian and Western European examples where standards were more lenient and emission levels higher. Thus emission levels would be higher in the absence of the program.

112. This survey was done for model years prior to 1981. "For 1981 and later model years, separate factors for vehicles with closed-loop system failures, misfueled vehicles and normal emissions control system operations are combined into aggregate emission factors based on estimated migration patterns from one category to another . . ." (Crandall et al., 1986, p. 93).

113. A pronounced deterioration of in-use automobile emissions equipment has been attributed to poor maintenance, component failure and tampering (Tietenberg, 1988). The situation could become far worse in the future with catastrophic failures of sophisticated control equipment. For evidence on the impact of such failures, see Crandall et al. (1986).

114. The problem of attributing air quality in a particular region to a particular emission reduction program goes beyond a lack of data. Varying meteorological factors from region to region affects how emission reductions are translated into ambient quality improvement. As well, there is a complicated nonlinear relationship between emissions of HC and NO_x and the creation of photochemical smog.

115. There is some evidence however, that the 31 percent improvement over this period is biased upwards (Crandall et al., 1986).

116. After comparing three U.S. regulatory programs related to the automobile-- fuel economy, safety and emissions regulations--Crandall et al (1986) conclude that the automobile emission control program ". . . is the most inefficient and poorly designed of the three regulatory programs analyzed . . ." (p. 157)

117. A two-car strategy has been suggested in the literature. One car would adhere to strict emission control strategies, the other would not. Only low emission cars would be registered in high pollution areas. It has been estimated that in the U.S. only 35 percent of the fleet would need to be low emitters (Tietenberg, 1988).

A two-car system already implicitly exists in the U.S., since California has its own emission standards. Of course the current situation does not yield significant cost savings since both cars are low emitters.

118. It is assumed that these standards are met immediately, although in reality it may take some time to achieve such standards.

119. Such a policy represents a change in philosophy since the owner of the car rather than the manufacturer effectively becomes responsible for the emissions from his or her car.

120. For more on the spatial dimensions of this problem see section III(B) and IV(C).

121. Dewees (1974) also argues that current standards impose higher marginal control costs on some vehicles than others because emissions vary with weight and transmission type. Thus the current scheme is not even cost effective when evaluated only within the context of the fleet.

122. Among the first proponents of a charge related to auto pollution emissions were a group of economists from the Rand Corporation (Baumol and Oates, 1979). Their suggestion, known as a smog tax, involves regular inspections of a car's exhaust and a resulting assignment of a smog rating. The car's smog rating would determine the price paid by the motorist for a gallon of gasoline. The motorist could reduce personal

tax payments by maintaining the car's pollution control equipment, reducing the mileage driven or retrofitting an older model car to reduce its emissions.

123. Mills and White (1978) suggested a similar externality pricing scheme.

124. It has been argued that some people may just buy a polluting vehicle and pay the charge. This is, as Dewees points out, principally an equity argument. To prevent the sale of uncontrolled cars, the scheme can be adjusted to set a prohibitive charge for emissions above some predetermined level.

125. Such monitoring is already required in non-attainment regions in the U.S.

126. In Norway and Sweden only catalyst cars receive a tax advantage.

127. Some people have extended this argument to conclude that the externality creating activity is also producing too much primary output. Baumol and Oates (1988, chapter 7) show that this may not be true if the usual convexity conditions are violated (see section III(E)) or if there is more than one externality (see footnote 29).

128. At Pearson International Airport in Toronto, traffic increased from 250,000 to 350,000 movements between 1985 and 1988.

129. For a review of the several scales on which noise can be measured, see Gillen and Levesque (1990, chapter 2).

Noise is measured in decibels (dB) which are an indication of intensity derived from a logarithmic transformation of sound energy (Starkie and Johnson, 1975). Noise emissions from individual aircraft (as well as the regulations referred to above) are measured in effective perceived noise decibels (EPNdB). This scale incorporates several modifications to the dB scale to account for how people perceive the noise from an aircraft during the entire flyby. In addition to aggregating this noise, the EPNdB scale also incorporates weights which account for the greater annoyance from high frequency noises and from pure tones.

While EPNdB is meant to represent the subjective annoyance produced by the operation of a single aircraft, it is necessary also to have a cumulative or ambient measure of noise produced by all activity over some time period. The Noise Exposure Forecast (NEF) is designed to represent cumulative noise levels during a 24-hour period at a particular site. Following Muskin and Sorrentino (1978) it is measured as:

$NEF_{hk} = EPNdB_{hk} + 10 \log [d_{hk} + 16.67 n_{hk}] - 88$

where: NEF_{hk} = noise exposure forecast magnitude generated by aircraft h along flight path k;

$EPNdB_{hk}$ = effective perceived noise level in decibels given aircraft class h and flight path k;
d_{hk} = number of flight operations of aircraft class h along flight path k during daytime hours;
$16.67n_{hk}$ = number of flight operations of aircraft class h along flight path k weighted to take account of disproportionate annoyance impact of night-time flights;
constant 88 = normalization factor computed by a procedure which causes the NEF measure to be clearly of a different magnitude from EPNdB and to set NEF levels above some "threshold" value at which ground noise exposure levels would not be significant.

Total NEF at a specified ground point is found by summing the operations of all aircraft classes h and flight paths k:

$$NEF = 10 \log \sum_h \sum_k \text{antilog}(NEF_{hk}/10)$$

Points of equal NEF around an airport are connected to form NEF contours.

130. Noise regulation in Canada is under the jurisdiction of the Minister of Transport. For a summary of Canadian, U.S. and various European management strategies, see Gillen and Levesque (1990).

131. Aircraft certified under the original Part 36 standards are called Stage 2 aircraft. Stage 1 aircraft were those not subject to the Part 36 standards. In 1977 Part 36 was amended to set more stringent certification standards for new Stage 3 aircraft.

132. A similar charge derivation is also presented in Alexandre, Barde and Pearce (1980).

133. Nelson also calculates an implicit marginal cost per EPNdB abated. Since the retrofit reduces, on average, the EPNdB from the B-707 from 116.8 to 104, the marginal cost (MC) is:

$$MC = \frac{P}{\Delta EPNdB} = \frac{231}{12.8} = \$18.05.$$

134. Nelson does suggest a small charge for noise emissions below the standard but this appears to play no important role in the scheme.

135. This scheme is similar to the safety valve suggested in Roberts and Spence (1976) and Spence and Weitzman (1978).

136. "The assumption of a linear relationship between NEF and willingness to pay over the range of NEF30 to NEF45 also appears justifiable from the results of annoyance surveys, which do not indicate any significant nonlinearities in the percentage of people annoyed." (Harrison, 1983, p. 54).

137. A linear damage function would favour an externality price over a standards scheme or marketable permits scheme (see section III(C)).

138. For a summary of this technique and others for determining the damages caused by noise see Gillen and Levesque (1990, chapter 3). Also a survey of noise and pollution-related housing-value studies can be found in Button (1990).

139. The reduction in a house value is not a social cost of noise, merely a reflection of the annoyance of noise. For a review of the practical problems associated with housing-price studies, see Harrison and Rubenfeld (1978).

140. This is the threshold value below which noise is no longer considered a nuisance.

141. Alexandre, Barde and Pearce (1980) attempt to implement the COWPS formula discussed above. Their presentation appears to be illogical or incorrect but they do point out the apparent inconsistency of this assumption with the presumed linearity of the damage function.

142. Operational changes may be limited by safety concerns. As well, even where this is not the case, actual monitoring of noise from individual flights would be required in order to make such behaviour feasible.

143. See section IV(E) for more on the comparison of the dynamic efficiency of standard and charge schemes.

144. Alexandre, Barde and Pearce (1980) develop a charge designed to compensate individuals who live near airports.

145. Baumol and Oates (1988) argue that this may result in entry into the "victim activity." In our example this would mean too many homes located close to airports.

146. Presumably if the victim rented the device, he or she would be ineligible for the compensation next period, and would be at F in future periods rather than E.

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